

Q
11
S85X
NH

ISSN 0038-3872

SOUTHERN CALIFORNIA ACADEMY OF SCIENCES

BULLETIN

Volume 119

Number 1



SMITHSONIAN INSTITUTE, COPY 2
SMITHSONIAN INSTITUTION 10TH
ACQUISITIONS/EXCHANGE NMNH # 25
ST. AND CONSTITUTION AVE. NW
WASHINGTON DC 20013

14

119(1) 1-48 (2020)

April 2020

Southern California Academy of Sciences

Founded 6 November 1891, incorporated 17 May 1907

© Southern California Academy of Sciences, 2020

2019-2020 Officers

David Ginsburg, *President*
Lisa Collins, *Vice-President*
Edith Read, *Recording Secretary*
Amber Brown, *Treasurer*
Kristy Forsgren, *Corresponding Secretary*
Daniel J. Pondella II and Larry G. Allen, *Editors - Bulletin*
Brad R. Blood, *Editor - Smilodon Newsletter*
Shelly Moore, *Webmaster*
Ted Stankowich, *Membership*

ADVISORY COUNCIL

Ralph Appy, *Past President*
Jonathan Baskin, *Past President*
Brad R. Blood, *Past President*
John H. Dorsey, *Past President*
Julianne Kalman Passarelli, *Past President*
John Roberts, *Past President*

BOARD OF DIRECTORS

2017-2020	2018-2021	2019-2022
Amber Brown	Ann Bull	Mia Adreani
David Ginsburg	Kristy Forsgren	Lisa Collins
Shana Goffredi	Kimo Morris	Julianne Passarelli
Gordon Hendler	Shelly Moore	Edith Read
Gloria Takahashi	Ted Stankowich	Marcella White

Membership is open to scholars in the fields of natural and social sciences, and to any person interested in the advancement of science. Dues for membership, changes of address, and requests for missing numbers lost in shipment should be addressed to: Southern California Academy of Sciences, the Natural History Museum of Los Angeles County, Exposition Park, Los Angeles, California 90007-4000.

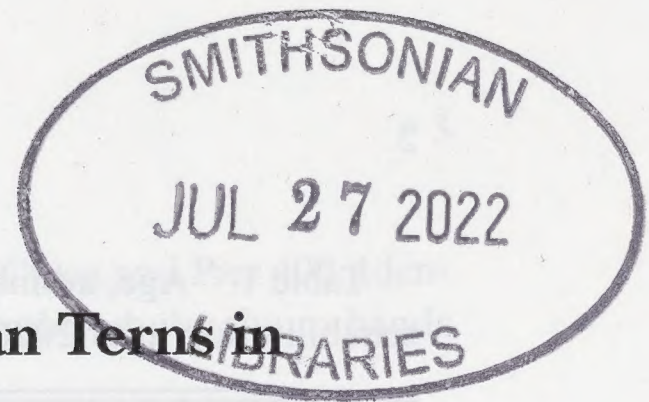
CATEGORY	E-BULLETIN ONLY	PAPER AND E-BULLETIN
Professional	\$60	\$80
Student*	\$10	\$30
Institutional	\$135	\$135
Life	\$2,500	\$2,500

* Must provide proof of student status with application

Fellows: Elected by the Board of Directors for meritorious services.

The Bulletin is published three times each year by the Academy. Submissions of manuscripts for publication and associated guidelines is at SCASBULLETIN.ORG. All other communications should be addressed to the Southern California Academy of Sciences in care of the Natural History Museum of Los Angeles County, Exposition Park, Los Angeles, California 90007-4000.

Date of this issue 26 May 2020



Survival and Recruitment of Rehabilitated Caspian Terns in Southern California

Julie M. Skoglund,^{1*} Rebecca S. Duerr,¹ and Charles T. Collins²

¹*International Bird Rescue, Los Angeles Center, 3601 South Gaffey Street, San Pedro, California 90751*

²*Department of Biological Sciences, California State University, Long Beach, California 90840-3702*

Every year thousands of birds are brought into care centers for rehabilitation in hopes that they can be treated and returned to the wild (McRuer et al. 2017). Many are victims of oil spills and other anthropogenic activities (Duerr et al. 2016; Henkel and Ziccardi 2018). In southern California, International Bird Rescue, Los Angeles (hereafter “IBR”) rehabilitates and releases 300 to 500 aquatic birds annually. Caspian Terns (*Hydroprogne caspia*) are among the many species of birds treated at IBR. Caspian Terns currently breed at three colony sites in southern California (Cuthbert and Wires 1999; Suryan et al. 2004; Collins 2006). Two of these sites—Pier 400 in the Port of Los Angeles, Los Angeles County (“Pier 400”), and Bolsa Chica State Ecological Reserve in northern Orange County (“Bolsa Chica”)—are approximately 18 km apart. The third is 180 km south on salt evaporation pond levees in the South San Diego Bay Unit of the San Diego Bay National Wildlife Refuge (“Salt-Works”) (Collins 2006). Since 2001, when IBR opened its Los Angeles facility, 69 Caspian Terns have been rehabilitated. Forty-four (63%) of these terns came from two separate incidents on barges anchored in Long Beach Harbor; nine others (13%) had injuries related to fish hooks and entanglement in fishing line. These barges presented an island-like habitat for nesting in an area that historically had islands available. Caspian Terns are known to have a prolonged post-fledging period during which young terns accompany their parents on foraging trips and migration (Cuthbert and Wires 1999), and thus it was not certain whether chicks rescued from these events and raised in captivity would survive and subsequently recruit into the breeding population.

The first barge incident took place in 2006 when an estimated 360 pairs of Elegant Terns (*Sterna elegans*) and 586 pairs of Caspian Terns nested on two barges anchored in the outer Long Beach Harbor (California Department of Fish and Wildlife, 2007 unpublished data). The barges’ owners directed their employees to wash the nests and small chicks overboard prior to moving the barges out of the area. This was an illegal activity, since these species are protected under the Migratory Bird Treaty Act of 1918 (the owners were successfully prosecuted and damages were assessed). Most of the chicks died; 413 dead chicks were recovered (California Department of Fish and Wildlife, 2007 unpublished data). Twenty-six chicks were rescued and taken to IBR for rehabilitation and release. One Caspian Tern chick was euthanized after 44 d in care because of an elbow infection; one Elegant Tern chick was euthanized on arrival because of a wing fracture. On average the chicks were in care at IBR for 46 d (range was 43–52 d). Nine Elegant Tern and 15 Caspian Tern chicks survived and were deemed strong enough to be safely released to the wild (Table 1). Since

* Corresponding author: Julie.Skoglund@bird-rescue.org

Table 1. Age, admission weight, and final disposition of Caspian and Elegant Tern chicks admitted for rehabilitation after 2006 and 2007 barge incidents.

Year, age	Admission weight (g)		Days in care		Disposition		N
	Mean*	Range	Mean	Range	Euthanized	Released	
Caspian Terns							
2006							
Downy	263.0	190-336	50.8	50-52		4	4
Pre-fledge	460.5	344-566	50.5	44-52	1	10	11
2007							
Downy	294.0		61			1	1
Pre-fledge	517.0	397-684	8.3**	1-32	2	25	27
Elegant Terns							
2006	178.4	74-216	41.5	1-57	1	9	10

* Adult body mass Caspian Tern: 538-782g (Cuthbert and Wires 1999).

** Mean shown includes birds kept longer due to injury.

adult Elegant Terns were still foraging in Los Angeles Harbor, the Elegant Tern chicks were released locally to join them prior to leaving on migration. Since most of the adult Caspian Terns had already departed, the Caspian Tern chicks were released at Salton Sea National Wildlife Refuge. The Salton Sea is an important migratory stop-over and foraging location for Pacific Coast Caspian Terns (Craig and Larson 2017).

The second barge incident took place in 2007 when a group of Caspian Terns nested on the flat deck of the Arctic Challenger, a large sea-going barge anchored in Long Beach Harbor (Ross 2008). The colony was estimated to be fewer than 100 pairs (Ross 2008), but it was not possible to get a definite count given that human disturbance can cause colony desertion (Cuthbert and Wires 1999). In late July 2007, several larger, nearly fledged chicks were found swimming near the barge. These had likely fallen off the flat edge of the deck, but numerous potential human disturbances that had taken place in the harbor near the barge during the nesting period may have contributed to these chicks falling into the water (Ross, 2008). Additionally, some chicks, having made a first flight off the barge, may have lacked the strength to return to the elevated deck and instead landed in the water nearby. Twenty-eight chicks were rescued and taken to IBR for rehabilitation. Chicks were deemed to be in releasable condition if in good body condition with blood values within normal limits and normal waterproof plumage, and if they were free from injuries or signs of disease. One chick with respiratory problems was euthanized. Most chicks were held at IBR for 2–5 d (Table 1), the intent being to reunite them with their colony at the barge and ideally with their parents. Four chicks were held longer at IBR because of age or injury, and were released at the Salton Sea in the same manner used in the 2006 release described above.

Twenty-three chicks were returned via boat to the vicinity of the barge over the course of several days while adults were still in the vicinity. All chicks being released were removed from transport kennels and held in hand to encourage vocalization in order to draw the attention of the adult terns on and near the barge. Once several adult terns were vocalizing and flying overhead, the chicks were allowed to fly away at will, and appeared to join the adults. It is unknown, but possible, that individual chicks were reunited with their parents or were adopted by other adults, a phenomenon reported for two other tern species

Table 2. Sightings of rehabilitated Caspian Terns at nesting colonies at Bolsa Chica and Pier 400. Identification of individual terns made by reading U.S. Fish and Wildlife Service numbered aluminum bands with a 20x-60x zoom spotting scope. Search effort varied by year.

Band number	2009	2010	2011-4	2015	2016	2017	2018	2019
925-76171	-	BC	-	-	-	-	-	-
925-76178	BC/P400	BC	-	-	-	-	-	-
1035-45297	-	-	-	-	-	P400	P400	P400
1035-45303	-	-	-	-	-	P400	-	-
1035-45312	-	-	-	P400	-	-	-	-
1035-45318	-	-	-	-	-	-	P400	-

BC = Bolsa Chica.
P400 = Pier 400.

(Saino et al. 1994). One of these chicks was discovered several days later with an injured carpal joint. An adult remained attentive by vocalizing and flying overhead until IBR staff removed the chick from the area. That injured chick was later euthanized. The ability of young Caspian Terns to dive and catch fish takes at least 60 d to develop (Cuthbert and Wires 1999). Juveniles tend to stay with, and be fed by, parents for several months (Cuthbert and Wires 1999). Accordingly, releasing young birds in the company of adults was thought to be important to their learning to forage for themselves and to follow on the subsequent migration to wintering areas. The 2007 juveniles that were released locally may have joined adults from the barge-nesting colony, and possibly their parents, before leaving on migration.

Although IBR bands most birds at release and has banded more than 24,000 rehabilitated birds since 2002, band recoveries are not common. An important question is how many rehabilitated birds survive and ultimately rejoin the breeding population. Several Caspian Tern chicks rehabilitated in 2006 and 2007 not only survived and rejoined a local breeding population, but also demonstrated subsequent nest-site fidelity (Table 2). Of the 15 chicks rehabilitated and released at the Salton Sea in 2006, two were seen breeding in 2009 and 2010 in the colony at Bolsa Chica (Table 2). One of these two was then observed at Bolsa Chica on 16 May 2009 and identified at Pier 400 on 25 June 2009. Such inter-colony movement of breeding adults between Bolsa Chica and Pier 400 was noted previously (Collins 2006). Of the 23 terns released in Long Beach Harbor in 2007, four have been sighted as adults breeding in the Pier 400 colony (Table 2).

Caspian Terns typically start breeding when three years old (Cuthbert and Wires 1999), but none of the 2007 chicks were observed until they were eight to ten years old. No searches for banded Caspian Terns have been made at Salt-Works since 2007. One additional encounter with a banded tern was discovered in an examination of North American encounter data for Caspian Terns provided by the USGS Bird Banding Laboratory. In this encounter (1035-45301), the bird was recovered on 23 May 2011 near Laguna Beach in southern Orange County after becoming entangled in fishing line. Unfortunately, the band was removed before the bird's release, so any affinity to one of the breeding colonies could not be determined. Annual survival of adult Caspian Terns is high, with some individuals reaching an age of >26 yr (Ludwig 1965). An adult originally banded as a chick at Bolsa Chica was observed at Pier 400 in 2017 at 23 years of age. However, the period of greatest mortality for Caspian Terns is the first six months of life (Ludwig 1942; Ludwig 1965), and only 57% of fledglings reach adulthood on the Pacific Coast (Gill and Mewaldt 1983).

We do not know which colonies were used by the unbanded barge-nesting adult terns in the years subsequent to 2006 and 2007. The fidelity of first-time breeding Caspian Terns to their natal colony is low (10%) (Cuthbert and Wires 1999). However, older adult Caspian Terns show substantial colony-site fidelity, with 69% being recorded at the same colony two years in a row, particularly if they bred successfully in the first year (Cuthbert 1988; Cuthbert and Wires 1999), and 29% breeding at nearby colonies. One of the 2006 chicks in this study bred in two successive years at Bolsa Chica, and one of the 2007 chicks bred in three successive years at Pier 400 (Table 2). The future survival and degree of colony-site fidelity of other 2007 individuals remain to be determined.

The observations reported here provide information on the survival of some of the Caspian Terns rehabilitated in southern California and their subsequent recruitment into a local breeding population. These observations also support high nest-site fidelity in this species. The rehabilitative efforts described were aided by several factors: the chicks entered care in sizable groups rather than as single chicks, many of the chicks were pre-fledglings so had spent several weeks with their parents, and the release locations and circumstances were very carefully chosen. Barges are not thought to be preferable to natural locations as nesting sites for Caspian terns, except possibly as a temporary option when new natural sites are being developed. The limited success in rearing young tern chicks in captivity, as documented here, does not suggest this is a viable alternative for augmenting populations of critically endangered species such as the Chinese crested tern (*Sterna bernsteini*). However, our results do suggest that, when properly conducted, ex situ conservation methods may prove successful in salvaging individuals that might otherwise perish because of an acute calamity at a critically endangered breeding colony. It remains possible that Caspian Terns may be more behaviorally flexible than previously thought with regard to acquiring adult life skills, but this may or may not be the case for other tern species.

Acknowledgments

Banding of the released terns was conducted under U.S. Fish and Wildlife Service Bird Banding Permit No. 21214 issued to Jay Holcomb of IBR and No. 22804 issued to Kathy Molina. Spencer Langdon, Kathy Keane, and the Port of Los Angeles made possible the observations of banded Caspian Terns at Pier 400. Peter Knapp contributed his observations of the Caspian Terns nesting at Bolsa Chica in 2009 and 2010 to this study. Kathy Molina banded chicks ready for release in 2006 and transported them from IBR to a release point at the Salton Sea. We are indebted to them both. Additional gratitude to the staff and volunteers at IBR, especially Jeri O'Donnell and Lauren Stoneburner.

Literature Cited

- Collins, C.T. 2006. Banding studies of Caspian Terns in southern California. *N. Amer. Bird Bander*, 31: 10–17.
- Craig, D.P. and K. Larson. 2017. Migratory connectivity of North American Caspian Tern (*Hydroprogne caspia*) populations. *Waterbirds* 40:58–62.
- Cuthbert, F.J. 1988. Reproductive success and colony-site tenacity in Caspian Terns. *Auk*, 105:334–344.
- Cuthbert, F.J. and L.R. Wires. 1999. Caspian Tern (*Sterna caspia*). In *The Birds of North America*, No. 403 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.
- Duerr, R.S., Ziccardi, M.H. and J.G. Massey. 2016. Investigations into mortalities during treatment: factors affecting oiled Common Murre (*Uria aalge*) survival to release through rehabilitation. *J. Wildl. Dis.* 52:495–505.
- Gill, R., Jr., and R.M. Mewaldt. 1983. Pacific coast Caspian Terns: dynamics of an expanding population. *Auk*, 100:369–381.

- Henkel, L.A. and M.H. Ziccardi. 2018. Life and death: how should we respond to oiled wildlife? J. Fish Wildl. Manag., 9:296–301.
- Ludwig, F.E. 1942. Migration of Caspian Terns banded in the Great Lakes area. Bird-Banding, 13:1–9.
- Ludwig, J.P. 1965. Biology and structure of the Caspian Tern (*Hydroprogne caspia*) population of the Great Lakes from 1896–1964. Bird-Banding, 36:217–233.
- McRuer, D.L., Gray, L.C., Horne, L., and E.E. Clark. 2017. Free roaming cat interactions with wildlife admitted to a wildlife hospital. J. Wildl. Manage., 81:163–173.
- Ross, W. 2008. Caspian Tern nesting on a barge in the Long Beach harbor, 2007 season. California Department of Fish and Wildlife, Wildlife Branch, Nongame Wildlife Program Report 2008-05. Sacramento, CA. 5 pp.
- Saino, N., Fasola, M. and E. Crocicchia. 1994. Adoption behaviour in Little and Common Terns (Aves; Sternidae): Chick benefits and parents' fitness costs. Ethology, 97:294–309.
- Suryan, R.M., D.P. Craig, D.D. Roby, N.D. Chelgren, K. Collis, W.D. Shuford and D. E. Lyons. 2004. Redistribution and growth of the Caspian Tern population in the Pacific Coast region of North America, 1981–2000. Condor, 106:777–790.

Mitigation Ponds Offer Drought Resiliency for Western Spadefoot (*Spea hammondi*) Populations

Katherine L. Baumberger,¹ Adam R. Backlin,¹ Elizabeth A. Gallegos,¹
Cynthia J. Hitchcock,^{1*} and Robert N. Fisher²

¹U.S. Geological Survey, Western Ecological Research Center, 1801 East Chestnut
Avenue, Santa Ana, CA 92701

²U.S. Geological Survey, Western Ecological Research Center, 4165 Spruance Road,
Suite 200, San Diego, CA 92101

Abstract.—Synergistic effects of habitat loss, drought, and climate change exacerbate amphibian declines. In southern California urbanization continues to convert natural habitat, while prolonged drought reduces surface water availability. Protection of biodiversity may be provided through mitigation; however, the long-term effectiveness of different strategies is often unreported. As a mitigation measure for building a new development within occupied *Spea hammondi* (western spadefoot) habitat in Orange County, California, artificial breeding pools were constructed at two off-site locations. *Spea hammondi* tadpoles were translocated from the pools at the development site to two off-site locations in 2005–2006. We conducted surveys a decade later (2016) to determine if *S. hammondi* were persisting and breeding successfully at either the original development site or the human-made pools at the two mitigation sites. We also verified hydroperiods of any existing pools at all three locations to see if any held water long enough for successful *S. hammondi* recruitment through metamorphosis.

During our study, no pooling water was detected at two of three main sites surveyed, and no *S. hammondi* were observed at these locations. Twelve of the 14 pools created at only one of the two mitigation sites held water for over 30 d, and we detected successful breeding at seven of these pools. Recruitment in some mitigation ponds indicated that *S. hammondi* habitat can be created and maintained over 10+ yr, even during the fifth year of a catastrophic drought. Therefore, this may also serve as a conservation strategy to mitigate climate change and habitat loss.

Global amphibian declines have been documented since the 1970s and are attributed to various causal factors (Blaustein and Wake 1990; Wyman 1990; Wake 1991; Drost and Fellers 1996; Fisher and Shaffer 1996; Grant et al. 2016), but habitat loss and degradation are cited as the most significant (Denton and Richter 2013; Thompson et al. 2016). In arid regions such as the US southwest, drought and climate change exacerbate declines, especially in regions with extensive urbanization (Diffenbaugh et al. 2015; Neal et al. 2018). The combination of these conditions can make life cycle completion challenging for pond-dwelling amphibians. Southern California is a region with high levels of urbanization that has experienced prolonged drought and is home to a suite of pond-dependent amphibians that have undergone major declines (Griffin and Anchukaitis 2014; Thomson et al. 2016).

In southern California there are four native species of anurans that use ponds or lentic pools for breeding (Fisher and Shaffer 1996). Of these, *Spea hammondi* is the only species

* Corresponding author: chitchcock@usgs.gov

that primarily uses non-permanent water sources for egg laying and larval development. Furthermore, ponds and vernal pools in this region are often isolated from one another, which makes amphibian populations less likely to recover if habitat is degraded and fragmentation becomes a problem (Denton and Richter 2013; Thomson et al. 2016). Restoration and mitigation can involve creating artificial ponds in adjacent or nearby similar habitat (Pechmann et al. 2001; Searcy and Shaffer 2008; Rannap et al. 2009). However, mitigation ponds/pools often do not mimic natural conditions and processes effectively and may not be successful (Lichko and Calhoun 2003; Arendt and Hoang 2005; Denton and Richter 2013). Conditions such as suitable hydroperiod, temperature, pH, and conductivity often cannot be replicated in artificial ponds but are important for breeding and developing amphibians, which frequently have specific thermal and chemical tolerances during their fully-aquatic stage. For instance, *S. hammondi* require a specific hydroperiod among other conditions for larval development, and this hydroperiod is less likely to occur during periods of drought.

A recent study using both mitochondrial and nuclear markers indicates that *S. hammondi* is comprised of two genetically distinct groups separated geographically by the Transverse Ranges of southern California (Neal et al. 2018). Though not designated as distinct species, the two clusters should be treated as separate conservation units as there is no current gene flow between them (Neal et al. 2018). Furthermore, the southern group has lost significantly more native habitat than its northern counterpart.¹ This study focuses on artificial habitats within the southern part of the range.

Spea hammondi has conservation status at several levels. The species is listed as Near Threatened by the International Union for Conservation of Nature,² under review for federal listing by US Fish and Wildlife Service (ECOS January 2020), a California Department of Fish and Wildlife (CDFW) Species of Special Concern, Bureau of Land Management (BLM) Sensitive Species (California Department of Fish and Wildlife 2019), and Natural Community Conservation Plan County of Orange Covered Species (Orange County NCCP/HCP 1995).³ *Spea hammondi* are small, nocturnal, burrowing anurans. The adults emerge from their burrows to breed during and immediately after rain events. Breeding typically occurs during a short window of time (2–3 wk) but has also been documented multiple times in one season, and time within-season might vary depending on weather patterns (Morey 2005; Morey and Reznick 2004; Ervin et al. 2005; Ervin and Cass 2007; Thompson et al. 2016). Historically, these anurans breed in vernal pools but are also known to take advantage of any seasonal water body, such as road ruts, cattle ponds, and artificial pools. Use of these anthropogenic habitats has been attributed to the fact that in southern California, more than 80% of historical *S. hammondi* habitat has been lost to development (Thompson et al. 2016).¹ Though the minimum time period for larval development in the laboratory was 14 d (Morey and Reznick 2004), in the wild pools must persist for a minimum of 30 d for *S. hammondi* larvae to complete development and metamorphose, regardless of habitat type (natural or anthropogenic; Morey and Reznick

¹ Jennings, M.R., and M.P. Hayes. 1994. Amphibian and reptile species of special concern in California. California Department of Fish and Game, Rancho Cordova, California 255pp.

² IUCN Red List. 2016. *Spea hammondi*. The IUCN Red List of Threatened Species 2016: <http://www.iucnredlist.org/>.

³ Natural Community Conservation Plan and Habitat Conservation Plan. County of Orange Central & Coastal Subregion Parts I & II: NCCP/HCP. 1995. Prepared for County of Orange Environmental Management Agency (December 7, 1995). 418 pp.

2004). Desiccation of larvae due to pools drying is frequent and widely documented for this species (Feaver 1971; Morey and Reznick 2004; Thompson et al. 2016).

Conservation planning for Orange County identified *S. hammondii* as a species requiring protection. Most of the remaining historical breeding pools in Orange County are on reserves or other protected open space. Three sites; (Irvine Mesa, Shoestring Canyon, and East Orange) are within reserves owned by Orange County (OC) Parks. They are located near the eastern side of OC in the foothills of the Santa Ana Mountains (Fig. 1).

Planned development was expected to affect ten *S. hammondii* breeding pools located at a site in East Orange. As mitigation to offset the development impacts, Glenn Lukos Associates, Inc., a private biological and wetland restoration consulting group, created 15 breeding pools at Irvine Mesa and six more pools at Shoestring Canyon, approximately 6.5 km to the northwest of Irvine Mesa (Fig. 1). *Spea hammondii* egg masses and larvae were translocated from the East Orange site to the mitigation pools at Irvine Mesa and Shoestring Canyon by Glenn Lukos in the winters of 2005 and 2006. Glenn Lukos Associates, Inc. monitored the translocation sites for five consecutive years to determine the success of the relocation. Although the development at East Orange never took place, Google Earth's satellite mapping shows that nine of the ten original natural pools were destroyed likely by agricultural disking sometime between September 2010 and March 2011.⁴

To determine if the mitigation ponds were sustaining *S. hammondii* 10 yr post translocation, we revisited the translocation sites in 2016. Sites were surveyed and assessed for the presence of the *S. hammondii* eggs, tadpoles, and adults at the mitigation ponds of Irvine Mesa and Shoestring Canyon. The original source ponds at East Orange were also assessed. The specific objectives of these surveys were to: 1) determine if *S. hammondii* were present at these sites, 2) determine which mitigation pools remained suitable for *S. hammondii* breeding, and 3) determine the hydroperiods of pooling habitat during the extreme drought.

Materials and Methods

The study area for this project included East Orange, Shoestring Canyon, and Irvine Mesa, within Orange County, California (Fig. 1). We determined *S. hammondii* presence and potential recruitment success using three methods: 1) nighttime visual encounter surveys for adults, 2) daytime visual encounter surveys for evidence of breeding (egg masses and larvae), and 3) monitoring of breeding pools until metamorphosis of the larvae. If pools could not be completely surveyed for larvae by visual inspection alone, we used dip nets to complement the survey. Three nighttime surveys were conducted at each site, targeting adult *S. hammondii*. On rainy nights we initiated surveys at the pools. Following an initial pool search, we walked spiraling outward around the pool at least 50 m into the terrestrial habitat, and searching continued between pools and throughout the site. We spent minimum of 4 hr searching each site during each night survey beginning about one half hour after sunset. We recorded all amphibian species encountered. All adult *S. hammondii* were weighed, measured, and their sex recorded. Additionally, versatile fairy shrimp (*Branchinecta lindahli*) were recorded when present. Daytime and nighttime surveys were conducted every 1–2 wk from 29 December 2015 to 28 April 2016, until *S. hammondii*

⁴ Google Inc. 2013. Google Earth (Version 7.1.2.20141) [Computer program]. Available at <https://www.google.com/earth/>. Accessed 18 April 2016.

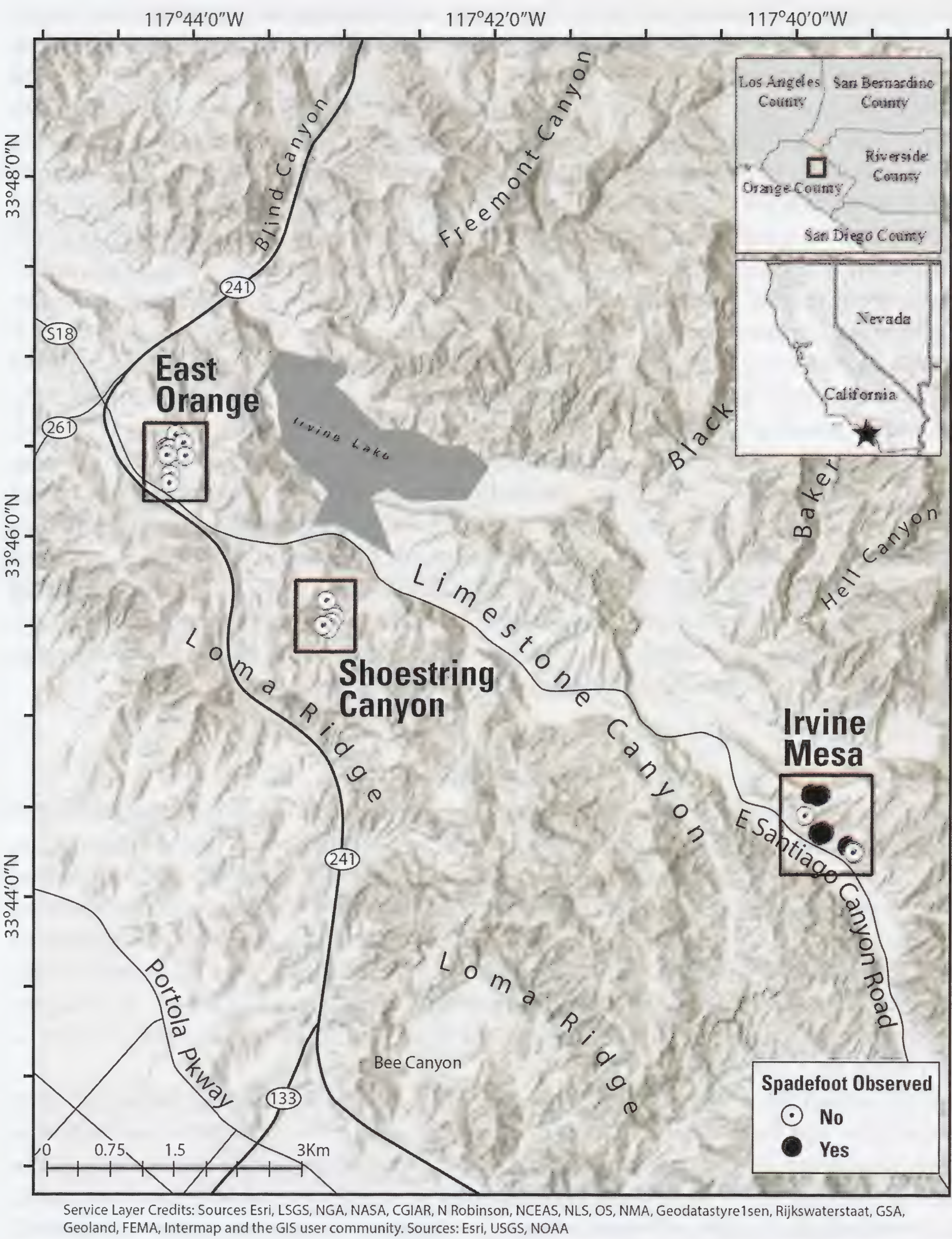


Fig. 1. *Spea hammondi* (western spadefoot) monitoring pools at East Orange donor site, and mitigation sites at Shoestring Canyon, and Irvine Mesa, 2016.

larvae metamorphosed or the pools dried. If the pool dried, monitoring ceased. If there was measurable rainfall afterwards, we resumed monitoring of that pool.

At each pool surveyed, we also recorded pool depth and basic water quality parameters (pH, conductivity, temperature). At pool locations that looked likely to hold water, we installed a water depth gauge and a Stream Temperature, Intermittency and Conductivity

(STIC) logger (Chapin et al. 2014). The water depth gauges we installed were simple 0.5 cm X 6.5 cm X 1.244 m fiberglass stream gauges (Forestry Suppliers Inc, Jackson, MS) fastened vertically to a ~0.6 cm X 61 cm rebar, which was pounded into the lowest point of the pool depression. The PVC had measurement lines at 2 cm intervals that were easy to read at a distance to record depth of water during each site visit. These provided a visual indication of pool depth throughout the breeding and larval development seasons. The STIC loggers are Onset Hobo Pendant temperature and light data loggers (Model UA-002-64) that have been modified to collect relative conductivity when submerged (Chapin et al. 2014). They can also be used to indicate water presence because when STICs are no longer submerged in water the conductivity will be zero (though this must be verified during surveys). Conductance was recorded via modification of the STIC light sensor, which is capable of measuring intensities between 0 and 330,000 lux. The light sensor was removed and two electrodes were attached to the open contacts to record electrical conductivity (EC) corresponding to this same 0–330,000 range, with 0 representing no water (Chapin et al., 2014). Therefore, STICs have a different electrical conductivity (EC) signal ranging from 0 to 330,000 (formerly lux, but as EC has undefined units), because they record relative conductivity (RC) not specific conductivity (SC). A mathematical conversion provided by Chapin et al. (2014) may be applied post hoc to convert the EC readings to specific conductance (SC) in μS . The STIC measurements were compiled and summarized in Microsoft Excel. We statistically compared pH, conductance, temperature, and hydroperiod between pools with tadpoles and pools without. We used 2-tailed t-tests for each water parameter. Computations were made using Microsoft Excel.

Results

The year 2016 was the fifth year of an extreme drought in southern California and a decade post translocation. In 2016 we surveyed all potential pools that remained from 2006 at the original East Orange location ($n = 10$), plus any newly discovered pools at that site in 2016 ($n = 4$). We also surveyed all pools created in 2006 at the mitigation sites, totaling seven at Shoestring Canyon and 14 at Irvine Mesa. We summarized which pools had *S. hammondi*, which had other anurans and fairy shrimp, what the average water temperatures were, and how long hydroperiods were at each pool (Tables 1–3). Many of the potential pools were simply dry depressions in the landscape that had the structure to hold water after rain events.

We identified five potential *S. hammondi* pools in 2016 at East Orange before the rainy season. One was from the original set of 10 pools surveyed in 2005–2006, and four were newly discovered on the property (Table 1). None held water during the winter of 2016 and no evidence of *S. hammondi* breeding was detected. We did not detect any adult *S. hammondi* during the surveys conducted at the East Orange site (Table 1), although other pond-breeding anuran species were detected.

None of the six mitigation pools at Shoestring Canyon held water during the 2016 rainy season. We found no evidence of *S. hammondi* breeding in any of the pools or in the nearby creek bed. We found no adult *S. hammondi* during the three nighttime surveys conducted at this site. In addition to the six pools built by Glenn Lukos Associates, Inc., we identified another possible pool behind an old berm (Pool 8), however no *S. hammondi* were detected there.

Twelve of the 14 mitigation pools at Irvine Mesa held water for >30 d. During our 2016 surveys, two of the mitigation pools built by Glenn Lukos Associates, Inc. merged

Table 1. Summary of *S. hammondi* (western spadefoot) observed (indicated as “X,” not observed indicated as “–”) at each monitoring pool in 2016 compared to translocations done in 2006.

Site	Pool	Spadefoot					Spadefoot Tadpole Desiccation 2016
		Spadefoot 2006	Egg Masses 2016	Spadefoot Tadpoles 2016	Spadefoot Metamorphs 2016	Spadefoot Adults 2016	
East Orange (1-10 were original, 11-14 were newly discovered in 2016)	1	–	–	–	–	–	–
	2	–	–	–	–	–	–
	3	–	–	–	–	–	–
	4	–	–	–	–	–	–
	5	–	–	–	–	–	–
	6	–	–	–	–	–	–
	7	–	–	–	–	–	–
	8	X	–	–	–	–	–
	9	–	–	–	–	–	–
	10*	X	–	–	–	–	–
	11	N/A	–	–	–	–	–
	12	N/A	–	–	–	–	–
	13	N/A	–	–	–	–	–
	14	N/A	–	–	–	–	–
Shoestring Canyon	1	X	–	–	–	–	–
	2	–	–	–	–	–	–
	3	–	–	–	–	–	–
	4	–	–	–	–	–	–
	5	X	–	–	–	–	–
	6	–	–	–	–	–	–
	8**	–	–	–	–	–	–
Irvine Mesa	1	–	X	X	X	–	–
	2	–	–	X	X	–	–
	3	–	X	X	–	–	X
	4	X	–	–	–	–	–
	6	X	X	X	X	–	–
	7	X	X	X	X	X	–
	8	X	–	–	X	–	–
	9***	X	X	X	X	X	X
	10	X	–	–	–	X	–
	12	X	X	X	–	X	–
	13	–	–	–	–	–	–
	14	X	X	X	X	–	X
	15	–	–	–	–	–	–
	16	–	–	–	–	–	–

* Pool 10 at East Orange was the only original pool not disked.
** Pool 8 at Shoestring Canyon was an additional pool. Not one of the mitigation pools built by Glen Lukos.
*** Pool 9 at Irvine Mesa combined with pool 5 and was counted as a single pool during the 2016 rainy season.

(Pools 5 and 9) and we considered them as one pool (Pool 9). We detected *S. hammondi* tadpoles in eight of the Irvine Mesa mitigation pools but documented successful breeding through metamorphosis at only seven of these pools in April 2016 due to desiccation and/or water quality (Table 1). *Spea hammondi* did not breed in all the pools with hydroperiods >30 d (Table 3). Newly metamorphosed frogs documented at Pool 8 may have originated from a nearby pool since we never detected any tadpoles in that pool. Pool 3

Table 2. Summary of other pond-breeding anurans plus fairy shrimp observed during 2016 surveys at each monitoring pool. (“X” indicates observed, “–” not observed).

Site	Pool	<i>Pseudacris hypochondriaca</i> (Baja California treefrog)	<i>Anaxyrus boreas</i> (western toad)	<i>Branchinecta lindahli</i> (versatile fairy shrimp)
East Orange	1	–	–	–
	2	–	–	–
	3	–	–	–
	4	–	–	–
	5	–	–	–
	6	–	–	–
	7	–	–	–
	8	–	–	–
	9	–	–	–
	10	–	–	–
	11	–	–	–
	12	–	–	–
	13	–	–	–
	14	–	–	–
Shoestring Canyon	1	–	–	–
	2	–	X	–
	3	–	–	–
	4	–	–	–
	5	–	–	–
	6	–	X	–
	8	–	–	–
Irvine Mesa	1	X	X	X
	2	X		X
	3	X	X	X
	4	–	–	–
	6	X	–	X
	7	–	–	X
	8	X	–	X
	9	–	X	X
	10	–	–	X
	12	–	–	X
	13	–	–	X
	14	–	–	X
	15	–	X	X
	16	–	–	X

dried before any of the tadpoles completed metamorphosis, and at Pool 12 the high pH and conductance readings we recorded could have contributed to recruitment failure (see below). We documented the presence of desiccated tadpoles at Pools 3, 9, and 14, but some individuals did metamorphose from Pools 9 and 14.

At Pool 12, we recorded a higher average pH (9.29) than at the other pools, and a higher average conductivity (169 μ S/cm) than at all pools but one (Pool 16, which had no tadpoles). The tadpoles in Pool 12 appeared smaller than those in other pools, and no metamorphosed *S. hammondi* were documented there. Pool 12 held water for 116 d, therefore hydroperiod was not a limiting factor (Morey and Reznick 2004). The pH at Pool 12 was significantly higher compared to all the other pools (mean = 8.05 $\pm$ 0.55 vs. mean = 9.29 $\pm$ 1.08; *n* = 9; *t* = –3.276; *p* = 0.01). When Pool 12 was excluded from the analysis,

Table 3. Summary of pool hydroperiods and water temperature for each monitoring pool, 2016.

Site	Pool	Consecutive Days Wet ¹			Water Temperature, °C ¹		
		1st Period ²	2nd Period	3rd Period	Average	Low	High
East Orange	1	N/A	—	—	—	—	—
	2	N/A	—	—	—	—	—
	3	N/A	—	—	—	—	—
	4	0	—	—	—	—	—
	5	N/A	—	—	—	—	—
	6	N/A	—	—	—	—	—
	7	N/A	—	—	—	—	—
	8	N/A	—	—	—	—	—
	9	0	—	—	—	—	—
	10	N/A	—	—	—	—	—
	11	0	—	—	—	—	—
	12	0	—	—	—	—	—
	13	N/A	—	—	—	—	—
	14	N/A	—	—	—	—	—
Shoestring Canyon	1	0	—	—	—	—	—
	2	0	—	—	—	—	—
	3	1	—	—	—	—	—
	4	0.2	—	—	—	—	—
	5	0	—	—	—	—	—
	6	0	—	—	—	—	—
	8	1	0.33	—	—	—	—
Irvine Mesa	1	132	10+	—	14.21	3.79	33.63
	2	113	—	—	13.75	4.41	33.95
	3	9	40	18	13.26	3.47	30.15
	4	2	11	3	11.9	1.76	27.96
	6	120	—	—	13.3	4.73	32.7
	7	118	—	—	14.08	4.21	32.39
	8 ³	120	—	—	N/A	N/A	N/A
	9	69	34	—	13.78	4.1	33.12
	10	51	20	5	12.57	2.41	28.36
	12	116	—	—	13.5	3.68	32.91
	13	19	3	—	10.87	4.52	21.57
	14	113	—	—	12.69	2.94	31.06
	15	3	48	17	13.59	6.27	29.95
	16	4	52	14	12.41	− 2.26*	31.27

¹ Data from STIC loggers installed in pools.
² Periods delineate the number of consecutive days a pool was wet.
³ STIC battery failed. Wet period is an estimate based on monitoring surveys.
* Pool was dry during this time. Temperature of the ground was below freezing.
N/A designates a pool where no STIC was installed. Pools that did not hold water have 0 consecutive days wet.

pH did not differ significantly between pools with tadpoles and pools without (mean = 8.10 ± 0.61 vs. mean = 7.85 ± 0.30; *n* = 26; *t* = 1.32; *p* = 0.20). Likewise, conductivity did not differ significantly for pools with tadpoles compared to pools without (mean = 113.49 ± 3896.18 vs. mean = 162.22 ± 13430.07; *n* = 20; *t* = −1.73; *p* = 0.10). Pool 16 had the highest average conductivity (293.33 μS/cm), and despite having a hydroperiod of 52 d, *S. hammondi* did not breed in the pool. Temperature was significantly

warmer for pools with tadpoles compared to pools without (mean = 13.57 ± 0.24 vs. mean = 12.27 ± 0.10 ; $n = 6$; $t = 2.73$; $p = 0.04$). Finally, hydroperiod was significantly longer for pools with tadpoles compared to pools without (mean = 104.56 ± 889.53 vs. mean = 36.20 ± 384.70 ; $n = 12$; $t = 5.16$; $p = 0.0003$).

Discussion and Conclusions

Many factors affect the availability of water for pond-breeding amphibians in southern California. The multi-year drought peaking between 2012 and 2015 affected the availability of vernal pool habitat for *S. hammondi* regionally, and climate change suggests unpredictable availability of surface water for the long term with the potential for more drought and anomalous weather patterns (Duellman 1999; Carey and Alexander 2003; Green 2016). Drought affects amphibian breeding in ephemeral systems more directly than in perennial ones since there are no alternative breeding locations within these systems (Miller et al. 2012). Though the planned development at East Orange did not occur, subsequent agricultural disking likely destroyed the vernal pool habitat there. Mitigation measures had already been put in place, and although many of the mitigation pools were unsuccessful in hosting *S. hammondi* breeding and larval development, seven of the pools held water long enough, with satisfactory thermal and chemical conditions, to allow breeding and recruitment for this species in 2016, 10 yr after mitigation. This result is promising not only for the populations that were translocated, but it has implications for pond-breeding species that are affected by climate change since we detected successful recruitment five years into a drought.

Even though we did not detect *S. hammondi* at the East Orange site during the winter of 2016, we cannot be certain this species has been eliminated. The 2015–2016 rainy season provided slightly more rain (24.82 cm) than the 16-yr average of 24.51 cm, but it may have been inadequate to fill the natural pools at East Orange. Online data show these rain events occurred during the appropriate time of year (December–May) to trigger breeding, however a recent study shows that late abundant rains may not be enough to trigger breeding if pools have not filled prior to these events (Shedd and Hansen unpub. data)⁵. For comparison, in the first year of Glenn Lukos Associates, Inc. translocation efforts (2005) the site received 46.61 cm of rain.⁵ Despite the destruction of the original pools, the East Orange site could still provide suitable breeding habitat for *S. hammondi* if there was enough rain to sufficiently saturate the soil and several restoration efforts were implemented. At East Orange, removing the drain at Pool 4 could facilitate water retention, and removing vegetation at Pools 12–14 could transform these depressions into functional *S. hammondi* breeding habitat. If subsequent surveys indicate that no *S. hammondi* persist at East Orange, repatriation of this species from the mitigation site at Irvine Mesa back to East Orange could be considered.

Though no water was present at Shoestring Canyon during our surveys, a small population of *S. hammondi* may persist there as well. *Spea hammondi* breeding was reported by consultants visiting the site at two pools in Shoestring Canyon during the 2009–2010 rainy season; however, biologists also reported that pools did not last long enough for the tadpoles to reach metamorphosis (T. Bomkamp and J. Meyer pers. comm). The longevity

⁵ California Department of Water Resources. 2016. California Data Exchange Center, Fremont Canyon weather station, Lat. 33.81110, Long. -117.70800. Available at http://cdec.water.ca.gov/cgi-progs/stationInfo?station_id=FMC. Accessed 11 April 2016.

of *S. hammondi* is not known though accounts of other spadefoot species estimate lifespans of up to 13 yr (Lannoo 2005). If adults are still present at Shoestring Canyon they could have bred in years when precipitation was timely and sufficient (Fisher et al. 2018). In winter of 2016 the pools at Shoestring Canyon were overgrown with vegetation, which may have contributed to their failure to hold water. To provide viable breeding habitat at this site, pools would likely have to be restored, including control of the invasive grasses around pools and in the surrounding upland habitat. Furthermore, the soil at the bottom of the pools may be too porous to hold water. Soil compaction and/or the installation of an artificial liner could facilitate water retention to provide adequate hydroperiod for *S. hammondi* to reach metamorphosis (T. Touré and T. Bomkamp pers. comm).

Although only four adult male *S. hammondi* were observed during surveys at Irvine Mesa, the many eggs and larvae at this site suggest that there is a persistent population here. Female anurans are generally less conspicuous than males, especially during the breeding season when males are calling (Thompson 2016), which is likely the reason we found eggs, larvae and metamorphs but only male adults. Additional surveys could provide an estimate of the population size and recruitment success. Data collected on pH, conductivity, temperature and hydroperiod at the pools having water show that only hydroperiod and temperature were significant predictors of spadefoot breeding. However, we had few pools with water, so a larger sample size would be desirable to help to reinforce or refute these data.

While most pools at the East Orange site were destroyed, and the region was experiencing unprecedented drought, the mitigation measures undertaken by Glenn Lukos Associates, Inc. in 2005–2006 established a population of *S. hammondi* at one of two mitigation sites. More remarkably, the population has persisted >10 yr post-translocation despite continued drought. With sufficient rain and continued protection of this habitat, the population at Irvine Mesa has the potential for recruitment adequate to provide individuals for re-patriation of the original East Orange site and possibly additional areas within the region. The long-term success of this translocation shows that mitigation pools can provide alternative breeding habitat for *S. hammondi* despite severe drought. Mitigation ponds may offer drought resiliency for unpredictable fluctuations caused by climate change.

Acknowledgements

We thank Zach Principe for bringing this project to fruition, our field crew Monique Wong and Jordyn Mulder, Kevin Neal for helping with field work, and our volunteers: Lily Sam, Brian Zitt, Adam Schroder, Stephanie Cashin, Jeff Ahrens, and Max Murray. We also thank Glenn Lukos Associates, Inc., especially T'Shaka Touré and Tony Bomkamp for their preliminary data on the sites and site history. We also thank the Irvine Ranch Conservancy, OC Parks and TNC for providing access to the sites. Funding was provided by The Nature Conservancy. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government. This is contribution number 725 of the U.S. Geological Survey Amphibian Research and Monitoring Initiative (ARMI).

Literature Cited

- Arendt, J., and L. Hoang. 2005. Effect of food level and rearing temperature on burst speed and muscle composition of western spadefoot toad (*Spea hammondi*). *Funct. Ecol.*, 19:982–987.
- Blaustein, A.R., and D.B. Wake. 1990. Declining amphibian populations: a global phenomenon? *TREE*, 5:203–204.

- Carey, C. and M.A. Alexander. 2003. Climate change and amphibian declines: is there a link? *Divers Distrib.*, 9:111–121.
- Chapin, T.P., A.S. Todd, and M.P. Zeigler. 2014. Robust, low-cost data loggers for stream temperature, flow intermittency, and relative conductivity monitoring. *Water Resour. Res.*, 50:6542–6548.
- Denton, R.D., and S.C. Richter. 2013. Amphibian communities in natural and constructed ridge top wetlands with implications for wetland construction. *J. Wildlife Manage.*, 77(5):886–896.
- Diffenbaugh, N.S., D.L. Swaina, and D. Toumaa. 2015. Anthropogenic warming has increased drought risk in California. *PNAS*, 112(13):3931–3936.
- Drost, C.A., and G.M. Fellers. 1996. Collapse of a regional frog fauna in the Yosemite area of the California Sierra Nevada, USA. *Conserv. Biol.*, 10:414–425.
- Duellman, W.E., ed. 1999. Patterns of distribution of amphibians. Johns Hopkins University Press.
- Environmental Online Conservation System (ECOS). 2020. Western spadefoot (*Spea hammondi*) species profile. Federal Register I, 59(219) Tuesday, November 15, 1994 I Proposed Rules. 58982–59028.
- Ervin, E.L., C.D. Smith, and S.V. Christopher. 2005. *Spea hammondi* (western spadefoot): reproduction. *Herpetol. Rev.*, 36:309–310.
- Ervin, E.L., and T.L. Cass. 2007. *Spea hammondi* (western spadefoot): reproductive pattern. *Herpetol. Rev.*, 38:196–197.
- Feaver, P.E. 1971. Breeding pool selection and larval mortality of three California amphibians: *Ambystoma tigrinum californiense*, Greay, *Hyla regilla*, Baird and Girard, and *Scaphiopus hammondi hammondi*, Girard. Masters Thesis, Fresno State College. 58pp.
- Fisher, R.N., C.S. Brehme, S.A. Hathaway, T.E. Hovey, M.L. Warburton, and D.C. Stokes. 2018. Longevity and population age structure of the arroyo southwestern toad (*Anaxyrus californicus*) with drought implications. *Ecol. Evol.*, 8(12):6124–6132.
- Fisher, R.N., and H.B. Shaffer. 1996. The decline of amphibians in California's Great Central Valley. *Conserv. Biol.*, 10(5):1387–1397.
- Grant, E.H.C., D.A. Miller, B.R. Schmidt, M.J. Adams, S.M. Amburgey, T.Chambert, S.S. Cruickshank, R.N. Fisher, D.M. Green, B.R. Hossack, P.T.J. Johnson, M.B. Joseph, T.A.G. Rittenhouse, M.E. Ryan, J.H. Waddle, S.C. Walls, L.L. Bailey, G. M. Fellers, T.A. Gorman, A.M. Ray, D.S. Pillod, S.J. Price, D. Saenz, W. Sadinski, and E. Muths. 2016. Quantitative evidence for the effects of multiple drivers on continental-scale amphibian declines. *Sci. Rep.*, 6:25625.
- Green, D.M. 2016. Amphibian breeding phenology trends under climate change: predicting the past to forecast the future. *Glob. Change Biol.*, 23(2):646–656.
- Griffin, D., and K.J. Anchukaitis. 2014. How unusual is the 2012–2014 California drought? *Geophys. Res. Lett.*, 41:9017–9023.
- Lannoo, M., ed. 2005. Amphibian Declines: The Conservation Status of United States Species. University of California Press.
- Lichko L.E., and A.J. Calhoun. 2003. An evaluation of vernal pool creation projects in New England: project documentation from 1991–2000. *Environ. Manag.*, 32(1):141–51.
- Miller, D.A.W., C.S. Brehme, J.E. Hines, J.D. Nichols, and R.N. Fisher. 2012. Joint estimation of habitat dynamics and species interactions: Disturbance reduces co-occurrence of non-native predators with an endangered toad. *J. Anim. Ecol.*, 81(6):1288–1297.
- Morey, S.R. 2005. *Spea hammondi* (Baird, 1859 “1857”), Western spadefoot. Pp. 514–517 in Amphibian declines: The conservation status of United States species (M.J. Lannoo, ed.). University of California Press.
- Morey, S.R., and D.N. Reznick. 2004. The relationship between habitat permanence and larval development in California spadefoot toads: field and laboratory comparisons of developmental plasticity. *Oikos*, 104:172–190.
- Neal, K.M., B.B. Johnson, and H.B. Shaffer. 2018. Genetic structure and environmental niche modeling confirm two evolutionary and conservation units within the western spadefoot. (*Spea hammondi*). *Conserv. Genet.*, 19(4):937–946.
- Pechmann, J.H., R.A. Estes, D.E. Scott, and J. Whitfield Gibbons. 2001. Amphibian colonization and use of ponds created for trial mitigation of wetland loss. *Wetlands*, 21(1):93–111.
- Rannap, R., A. Lõhmus, and L. Briggs. 2009. Restoring ponds for amphibians: a success story. *Hydrobiologia*, 634:87–95.
- Searcy, C., and H.B. Shaffer. 2008. Calculating biologically accurate mitigation credits: Insights from the California tiger salamander. *Conserv. Biol.*, 22(4):997–1005.

- Thomson, R.C., A.N. Wright, H.B. Shaffer, B. Bolster, K. Cripe, S.J. Barry, R.N. Fisher, and H.H. Welsh. 2016. California Amphibian and Reptile Species of Special Concern. University of California Press. 408 pp.
- Wake, D.B. 1991. Declining amphibian populations. *Science*, 253(5022):860.
- Wyman, R.L. 1990. What's happening to the amphibians? *Conserv. Biol.*, 4(4):350–352.

Are Fishes Attracted to Piers? Movements and Association of Marine Fishes to a Public Fishing Pier within a Commercial Harbor

Armand A. Barilotti,* Connor F. White, and Christopher G. Lowe

Department of Biological Sciences, California State University Long Beach, Long Beach, CA, 90840

Abstract.—Ocean fishing piers are ubiquitous along the world’s coastline, yet little research has examined how these structures can attract and retain fishes. Fishers routinely use these human-made structures as a reliable way to catch fish for subsistence or recreation. California halibut (*Paralichthys californicus*) and white croaker (*Genyonemus lineatus*) are commonly caught from fishing piers in southern California; however, some individuals have been found to contain high concentrations of hazardous contaminants. Thus, human health hazard warnings are posted throughout the Los Angeles area to limit fish consumption. To document attraction, residency, and association to fishing piers, 42 California halibut and 198 white croaker were tagged with acoustic transmitters in regions of the Los Angeles and Long Beach Harbors, including a local fishing pier, and the movements of these fish were tracked throughout a 1.5 year period. Average ($\pm$ SD) fish residency near piers was 90.5 ± 104.8 days for California halibut and 31.9 ± 25.7 days for white croaker. Only 18% of white croaker and 6% California halibut were detected migrating to the pier from other locations of the LA-LB Harbors, and most spent < 10 min within 300 m of the public fish pier. Only 14% of California halibut and 0.35% of white croaker geo-positions were within casting range (approximately 30 m) of the pier, thus California halibut show the greatest potential affinity for pier habitat. Due to their movement patterns and habitat associations California halibut are much more likely to be attracted to fishing piers than white croaker.

Piers are common structures found throughout coastlines and embayments of the world; surprisingly, little research has examined how fish utilize these structures, even though they have the potential to function similarly to artificial reefs. Artificial reefs are human-made structures often designed to simulate reef structure to support fishes and other marine life, thus have been implemented for remediation of damaged reefs or to increase fish stocks (Alevras and Edwards 1985; Baine 2001; Bohnsack 1989; Deysher et al. 2002; Schroeter et al. 2015). Most are constructed to provide complex habitat often including vertical relief, hard substrata, high rugosity, etc., which are thought to be attractive features for a variety of fishes (Baine 2001). Though mainly constructed to support industrial, commercial, or recreational purposes, pier pilings and support beams provide complex habitat, which can serve to unintentionally attract fishes to these structures. The pier pilings provide hard substrata, which fosters invertebrate and algae recruitment (Butler and Connolly 1999; Keough 1983), and provide both food and protection for a variety of fishes. They also offer vertical relief and shade which has been correlated with an increase of abundance of forage fish at piers (Able et al. 2013). Vertical habitat relief in other marine systems has been linked

* Corresponding author: aabarilotti@gmail.com

with increased abundances, recruitment, and diversity of fishes (Carr 1991; Caselle et al. 2002; Kellison and Sedberry 1998; Martin and Lowe 2010; Rilov and Benayahu 1998). Piers in southern California are generally located in areas containing a homogenous flat seascape of mud/silt or sand, offering increased foraging opportunities for fishes that utilize ecotone environments. The ecotone between a rocky reef and sand has been found to contain greater diversity and abundance of epi and infaunal species than the communities found in the sediments further from the reef (Alongi 1989; Ambrose and Anderson 1990; Barros et al. 2001).

Public ocean and bay fishing piers have been a part of California's history for over a century and have attracted both recreational and subsistence fishers due to their ease of access, low cost, no required fishing license¹, and relatively consistent fish productivity (Jones 2004). Unfortunately, the majority of public piers in southern California (San Clemente to Ventura) lie within a region where fish consumption warnings have been enacted to limit the consumption of many potentially contaminated coastal fish species². Cabrillo Pier is located in the center of this consumption warning zone, and is the closest pier to the EPA superfund site on the Palos Verdes Shelf (PV) as well as contaminated regions of the Los Angeles and Long Beach Harbors (henceforth referred to as LA-LB Harbors) (Fig. 1). Two contaminants of particular concern are types of organochlorines, dichlorodiphenyltrichloroethane (DDT) and polychlorinated biphenyls (PCBs). DDT was once a heavily used pesticide, and PCBs were an industrial chemical used in many applications; both are known carcinogens, and endocrine and reproductive disruptors (Longnecker et al. 1997). From the 1940's through early 1970's these chemicals were discharged into the local wastewater treatment system and watersheds. Since DDT and PCBs are resistant to degradation, they have persisted in the environment where they are bioaccumulated in the tissues of organisms and biomagnified through the food web due to their lipophilic properties (Brown et al. 1998; Malins et al. 1987; Zeng and Venkatesan 1999). Through these processes, many of the top marine predators within the Southern California Bight have been found to contain high tissue levels of organochlorines (Blasius and Goodmanlowe 2008; Mull et al. 2013).

Due to the proximity of the public Cabrillo Fishing Pier in LA Harbor to areas of relatively high sediment contamination, there is a concern that fish caught from the pier may contain harmful amounts of contaminants. Indeed, some fish caught and sampled near the pier have been found to contain high concentrations of organochlorines in their tissues; however, these concentrations vary widely and are likely attributed to the degree to which some individuals move and the habitats in which they feed (Anderson et al. 2001). Two species of fish that are commonly caught and consumed from coastal piers are white croaker (*Genyonemus lineatus*) and California halibut (*Paralichthys californicus*). They were chosen for this study because while they are benthic soft-substrata associating species, they have differences in habitat use, foraging ecology, and trophic position. In addition, in California white croaker is considered a "sentinel species" for contaminant exposure (Love et al. 1984; Malins et al. 1987). Both are known to obtain high concentrations of organic contaminants (84.9-2520 ppb DDT and 58.5-279 ppb PCBs for white croaker, 35.4-171

¹ California Fish and Game Code, section 7153.

² Office of Environmental Health Hazard Assessment - <https://oehha.ca.gov/media/downloads/advisories/socaladvisoryl61809.pdf>



Fig. 1. Map of capture and release locations of passively tracked white croaker and halibut. Green dots indicate release locations of white croaker and blue dots represent locations of release for halibut. The red diamonds represent VR2W receiver locations and the beige circles around them are the nominal detection range of the receivers.

ppb DDT and 5.61-25 ppb PCBs for California halibut³). Since sediments from around Cabrillo Pier have been found to contain relatively low levels of organochlorine contaminants, it is likely some fish are being attracted to Cabrillo Pier from other more contaminated locations like the PV Shelf, Consolidated Slip (LA Harbor), Fish Harbor (LA Harbor), or inner LB Harbor (Anderson et al. 2001; Eganhouse and Pontolillo 2008). To identify the most important areas for sediment remediation, it is critical to determine which contaminated regions are the most connected to Cabrillo Pier.

Though both species are demersal soft substratum associated, they utilize these environments differently due to their foraging strategies, thus could be attracted to Cabrillo Pier based on different habitat attributes. White croaker are roving benthic foragers, selecting habitat that contains fine grain sized sediment with high levels of total organic carbon and high densities of benthic infauna, which describes the habitat surrounding Cabrillo Pier (Ahr et al. 2015; Love et al. 1984). Unlike white croaker, California halibut are lay-in-wait ambush predators of bait fish (e.g. *Engraulis mordax*, *Sardinops sagax*, *Leuresthes tenuis*) and epibenthic invertebrates, often associated with the ecotone between sand and reef or eelgrass (Freedman et al. 2015; Freedman et al. 2016; Haaker 1975). The ecotone, pier

³ National Oceanic and Atmospheric Administration- https://19january2017snapshot.epa.gov/www3/region9/superfund/pvshelf/pdf/montrose_report.pdf

pilings, and vertical relief are attractive habitat features for bait fishes, thus the area near Cabrillo Pier could increase foraging opportunities for California halibut (Able et al. 2013; Johnson et al. 1994). Since Cabrillo Pier has potentially attractive habitat features for both species, the first goal of this study is to quantify the probability of both species moving to Cabrillo Pier from other areas of the LA-LB Harbors. In addition, test the hypotheses that 1) white croaker and California halibut exhibit selection for the habitat in close proximity to Cabrillo Pier; and 2) individuals visiting Cabrillo Pier from other regions of the LA-LB Harbors will display a similar degree of site fidelity to Cabrillo Pier as those tagged near Cabrillo Pier.

Materials and Methods

This study was conducted in the LA-LB Harbors complex located in southern California, USA (Fig. 1). White croaker and California halibut were caught and tagged throughout seven regions of the LA-LB Harbors. Regions were designated based on habitat differences and geospatial boundaries in order to compare fish movements across habitat types. These regions were Los Angeles outer harbor (LAOH), the Los Angeles inner harbor (LAIH), the Long Beach inner harbor (LBIH), the Long Beach outer harbor (LBOH), Fish Harbor, Cabrillo Pier, and Pier J, and the area off White Point, Palos Verdes (PV Shelf) (Fig. 1). They were chosen to examine fish movement within and among regions of the LA-LB Harbors, especially between regions with high sediment contamination and regions with public fishing piers.

Cabrillo Pier is located in the southwestern region of the LA-LB Harbors and is a public fishing pier that runs parallel to the Federal Breakwater and is 375 m long by 5-10 m wide. Water depth directly below the pier ranges from 1 m at northwestern end to 3 m at the southeastern end. The pier rests on the edge of a sand/mud shelf, which drops off from the pier to a seafloor depth of approximately 7 m. The substrata around Cabrillo Pier consists of mud, silt, and sand. There is a low-lying rock reef, partially buried under unconsolidated sediments at the eastern edge of the pier that runs perpendicular to the pier.

To characterize the movements of white croaker and California halibut throughout the LA-LB Harbors and at fishing piers, 240 fish were caught and surgically fitted with acoustic transmitters and then passively tracked from July 2013 to January 2015. At least 25 white croaker and 2 halibut were tagged within each of the regions of the LA-LB Harbors, and 9 white croaker were tagged on the PV Shelf. All white croaker were caught using hook and line, and California halibut were caught using hook and line or a 3 m wide otter trawl. Fish were anesthetized in a bath of Tricaine methanesulfonate (MS222) (0.2 g/L of seawater) until reaching stage-4 level of anesthesia (Freedman et al. 2015; Wolfe and Lowe 2015). Weight (g), standard length (mm), fork length (mm), and total length (mm) were recorded (Summerfelt and Smith 1990). All fish were surgically fitted with a Vemco V9-1X acoustic transmitter (24 mm x 9 mm, nominal pulse interval 120-250 s, power output 145 dB, battery life 363 d) coated with a 1:2.3 mixture of beeswax and paraffin to reduce an immune response (Lowe et al. 2003). A 2 cm incision was made through the abdominal wall and the acoustic transmitter was inserted into the peritoneal cavity of the fish and closed with 2 sutures of dissolvable monofilament suture (Ethicon, 5-0 PDS*II). Once the fish recovered from surgery (minimum of 5 min) it was released near its capture location. California halibut were also externally tagged with a numbered spaghetti tag in the dorsal fin musculature. All capture, handling, and surgical procedures complied with the California State University Long Beach's IACUC #325.

To document fish movement throughout the LA-LB Harbors, 38 VR2W (Vemco, Ltd.) omni-directional underwater acoustic receivers were positioned at key locations around the LA-LB Harbors and surrounding areas from July 2013 through January 2015 (Fig. 1). Receivers were placed approximately 1-2 m off the seafloor at choke point areas to function as gates to monitor fish movement from different areas of the LA-LB Harbors. All receivers were deployed in July 2013, except for the two receivers placed in Fish Harbor (deployed August 2013 and November 2013), and the four additional receivers deployed at Cabrillo Pier in December 2013 to construct the Vemco Positioning System (VPS) array (Fig. 1). The VPS array was used to obtain fine-scale geo-position estimates of tagged fishes based on trilateration when a transmission was detected by three or more acoustic receivers (Espinoza et al. 2011; Wolfe and Lowe 2015). VPS receivers were paired with a synchronization transmitter (V16-5x, 69 kHz, 300 s pulse interval, power output of 150 dB) to correct for clock drift and improve positional accuracy (Espinoza et al. 2011). Receiver maintenance and downloads were conducted bimonthly. Mean maximum detection range for receivers at Cabrillo Pier was determined to be 401 ± 19 m (mean $\pm$ SD) (Fig. 1). Detection range of other receivers was used from previous studies (Farris et al. 2016; Wolfe and Lowe 2015). Fish detections from VR2W receivers were downloaded and managed using VUE v 2.2.2 (Vemco). Data analyses were carried out using R 3.0.2 (R Foundation for Statistical Computing) and PRIMER-E v.6 (Plymouth, UK) and maps were created using ArcGIS 10.2 (ESRI).

The acoustic receiver detection dataset was filtered to remove false detections and detections from individuals that were thought to have died. If a transmitter was detected continuously by the receiver closest to the fish's release location for the entire battery life of the transmitter, the fish was classified as a mortality from tagging stress. To determine potential predation of tagged fish, the displacement speed of movements between two receiver locations were calculated. White croakers have been estimated to have a maximum sustained swimming speed of 0.61 m/s (Dorn et al. 1979), therefore any tagged white croaker that were observed to travel faster than 0.61 m/s between receivers were considered predation events. Maximum sustained swimming speeds of California halibut are unknown so this test could not be performed for the halibut detection data; however, no erratic or excessively rapid ($> 1 \text{ m s}^{-1}$) movements were documented for California halibut that would be characteristic of a predation event.

Several receivers had overlapping detection ranges allowing an individual to be simultaneously detected at multiple receivers. All receivers within each one of these groups were aggregated and treated as a single station (PV curtain, Angel's Gate, Queen's Gate, Pier J, Consolidated Slip, Fish Harbor and Cabrillo Pier). Furthermore, within each of these receiver groups, any detection from an individual that occurred less than 120 s (minimum pulse interval) after a previous detection was removed, as these represented simultaneous detections on multiple neighboring receivers. Removing these duplicated detections prevented artificially inflating the number of detections within a region.

Two methods were used to determine the likelihood of white croaker and California halibut moving from other regions of the LA-LB Harbors to Cabrillo Pier. First, was calculating the proportion of individuals tagged in each region of the LA-LB Harbors that were detected at Cabrillo Pier. Second, a one-way analysis of similarity (ANOSIM) was used to compare patterns of receiver presence across individuals that were tagged in different regions of the LA-LB Harbors (Garcia et al. 2015; Meyer et al. 2010; Papastamatiou et al. 2015). Similarities were based on daily receiver presence, which was calculated as the cumulative number of days an individual was present (> 1 detection within a day) for each

receiver. A dispersion weighting transformation was used to normalize for high variability in daily receiver presence among individuals from each region of the LA-LB Harbors (Clarke 2014). The degree of overlap (R-value: < 0.25 = high overlap, > 0.75 = low overlap) was used to evaluate receiver usage between groups tagged in different regions of the harbor (Garcia et al. 2015; Meyer et al. 2010). Pair-wise comparison significance ($p < 0.05$) was interpreted as fish tagged in different regions primarily using a different core cluster of receivers. Non-metric multidimensional-scaling ordination (nMDS) plots were used to visualize the relative similarities of receiver presence among individuals from regions of the LA-LB Harbors.

Euclidean Distance-based Analysis (EDA) was used to quantify the pattern in spatial association between individuals and the pier. For each individual, the distance between each VPS rendered position and the closest edge of the pier was measured (Conner and Plowman 2001). These values were then compared to the available habitat, which was calculated as an equal number of randomized positions distributed within the detection range (approx. 250 m) of the Cabrillo Pier array. Distances from the pier to the randomized and VPS rendered geo-position of tagged fish were placed into 10 m bins for white croaker and 5 m bins for California halibut, as they had a larger number of VPS detections. To determine if the pier was influencing the species distributions within the Cabrillo Pier VPS array, the counts of the VPS and randomized positions were compared using a Pearson's chi-square test. Individual's pier selection index was calculated as the ratio of number of observed positions to randomized positions in each distance bin. The pier selection index was interpreted as values > 1 representing selection/association, values < 1 representing non-selection. Pier selection indices were averaged across individuals to create a species-wide pier selection index. Pier association for either species was defined as having a pier selection index > 1 within 50 m of Cabrillo Pier. In addition, there is a low-relief rocky reef that runs perpendicular to the southeast corner of the Pier, to determine if tagged individuals additionally used ecotone provided by this reef an EDA analysis was conducted to compare distances used to this reef.

Some California halibut had periods where they were detected consistently at a high frequency and some individuals had orders of magnitude more rendered VPS positions than others, allowing for temporal autocorrelation or a subset of individuals to drive results. To reduce effects of potential biases a bootstrapping procedure was used where 100 positions were randomly sampled from each individual with replacement and pooled across individuals (Teesdale et al. 2015). This process was repeated 1000 times and compared to the null model using a χ^2 test. This analysis was not used for white croaker due to low sample size and number of VPS rendered positions.

While the pier can impact what habitat individuals' use, it can also impact their fidelity to the area. The degree of site fidelity of fish caught and tagged in close proximity to Cabrillo Pier was compared with individuals tagged at other locations throughout the LA-LB Harbors that visited the pier, using two metrics: daily presence and duration at Cabrillo Pier. Daily presence was the sum of the days a fish was detected at Cabrillo Pier while the duration spent at Cabrillo Pier was determined by multiplying the average transmitter pulse interval (182 s) by the total number of detections at the pier for each fish. This time (duration) was then divided by an individual's daily presence, which yielded an average number of hours per day an individual was detected at the pier receivers.

A non-parametric randomization test was used to compare these site fidelity metrics between fish tagged at Cabrillo Pier and those visiting from other regions of the LA-LB Harbors. This method combined all individuals' fidelity data into one pool, and sampled

Table 1. Average total length (mm) and number (n) of white croaker (WC) and California halibut (CH) tagged by region. First two columns after Region are of white croaker and the last two columns are of California halibut.

Region	WC mean TL (mm) ± SE	n	CH mean TL (mm) ± SE	n
Cabrillo Pier	228 ± 4.2	29	455 ± 17.3	26
Fish Harbor	229 ± 2.1	30	NA	0
LAIH	252 ± 3.5	26	486 ± 64.3	4
LAOH	233 ± 1.9	29	492 ± 48.2	2
LBIH	237 ± 2.1	25	592 ± 20	2
LBOH	224 ± 2.1	25	416 ± 22.5	2
Pier J	231 ± 2.2	25	538 ± 72.5	6
PV Shelf	235 ± 3.4	9	NA	0

with replacement to generate two distributions that equaled the sample size of fish tagged at Cabrillo Pier and fish visiting from other regions. The difference in mean value of the two samples was calculated and repeated 10,000 times to create a distribution of expected differences between the means assuming all individuals of a species belonged to one population that utilized Cabrillo Pier equally. The observed difference to the created null distribution generated the probability of observing this outcome if all fish utilized Cabrillo Pier similarly. California halibut were not compared due to low sample size of individuals visiting from other regions of the LA-LB Harbors.

Results

A total of 42 California halibut and 198 white croaker were caught and tagged at locations throughout the LA-LB Harbors (Fig. 1, Table 1). White croaker averaged 234 ± 1.1 mm total length (TL), and all individuals tagged were larger than size at 100% maturity (190 mm TL) while California halibut averaged 472 ± 15 mm TL (mean $\pm$ SEM) (Love et al, 1984). Sex could not be determined for most individuals of either species due to the lack of sexual dimorphism. Only one California halibut (Transmitter ID 11392; 615 mm TL) was reported as being recaptured by a fisher, and it was released after being measured.

All 38 receivers recorded detections of tagged fish, and mean ($\pm$ SEM) number of detections was highly variable among the two species ($14,708 \pm 31,869$ for California halibut, $6,202 \pm 8,073$ white croaker). Twenty-seven white croaker and one California halibut were never detected on any receiver, and these individuals were excluded from all movement analyses. Many of these fish were not released in the vicinity of a receiver (> 300 m); however, most were released inside the LA-LB Harbor and therefore should have had a greater likelihood of detection. None of the nine white croaker tagged on the PV Shelf were detected by any of the 38 acoustic receivers used for this study. The proportion of white croaker identified as having died as the result of capture, handling, and surgery (tagging mortality) was 3.5% (7 individuals). Approximately 11.5% (23 individuals) of white croaker were estimated to have been eaten by a predator based on rate of movement between receivers (> 0.61 m s⁻¹). No California halibut were observed to exhibit rapid movements between receivers that could be considered a predation event. Four California halibut were detected making > 40 km movement from the LA-LB Harbors to receivers in Santa Monica Bay. A total of 163 white croaker and 41 California halibut were used for movement

analyses after removing individuals that were never detected or were determined to die from tagging mortality.

Receivers in the Cabrillo Pier VPS array detected a total of 58 individual white croaker and 27 California halibut. 28 of the white croaker were tagged in the vicinity of Cabrillo Pier and the other 30 individuals were tagged in other regions of the LA-LB Harbors. Of the white croaker visiting Cabrillo Pier from other regions, most were from LAOH, but 12-17% of fish detected were tagged in Fish Harbor, LBIH, and LAIH (Table 2). Twenty-seven California halibut were detected by the Cabrillo Pier array, and of those, 26 were initially tagged at Cabrillo Pier (Table 2).

White croaker tagged in different regions of the LA-LB Harbors did not show similarity in daily receiver presence (ANOSIM, $R = 0.935$, $p = 0.001$, Table 3, Fig. 2A). Pair-wise comparisons revealed that white croaker tagged in different regions of the LA-LB Harbors showed low overlap in daily receiver presence, $R > 0.75$ for almost all regions, except between individuals from Cabrillo Pier and LAOH ($R = 0.646$), and LBIH and LBOH ($R = 0.69$), where there was a moderate amount of overlap between daily receiver presence. California halibut tagged in different regions of the LA-LB Harbors did not show similarity in daily receiver presence (ANOSIM, $R = 0.826$, $p = 0.001$, Fig. 2B).

A total of 563 fine-scale geo-positions were rendered for five white croaker within the Cabrillo Pier VPS array (Fig. 3). The distances of these positions from Cabrillo Pier was significantly different from a random distribution ($\chi^2 = 109.8$, $df = 34$, $p < 0.001$). The pier selection index of white croaker VPS-rendered positions to Cabrillo Pier exhibited one distinct peak between 50 to 120 m from Cabrillo Pier and two smaller peaks at approximately 220-230 m and 270-280 m from the pier (Fig. 3A). The distances of these positions from the adjacent low-lying reef was also significantly different from a random distribution ($\chi^2 = 194$, $df = 66$, $p < 0.001$). The reef selection index of white croaker positions to the reef had a peak from 0-20 m from the reef and multiple peaks 300-570 m from the reef (Fig. 3B).

A total of 46,615 fine-scale geo-positions were rendered for 11 California halibut within the Cabrillo Pier VPS array (Fig. 4). The distribution of distances of these points from Cabrillo Pier was significantly different from random distribution ($\chi^2 = 170146.1$, $df = 71$, $p < 0.001$). The pier selection index suggests that California halibut mostly selected for habitats approximately 25-130 m from Cabrillo Pier when within 300 m of the pier (Fig. 4A). The distances of these positions from the low-lying reef was also significantly different from random distribution ($\chi^2 = 11120$, $df = 127$, $p < 0.001$). The distribution of reef selection index shows that California halibut selected for areas within 25 m from the reef, and sporadically > 45 m from the reef (Fig. 4B). California halibut pier and reef selection indices were robust against unbalanced sample size and temporal autocorrelation ($p < 0.001$) for all 1000 bootstrapped permutations.

White croaker caught and tagged in the vicinity of Cabrillo Pier spent an average of 31.9 ± 25.7 (mean $\pm$ SD) days at Cabrillo Pier, which was significantly greater than the total number of days spent at the pier by fish tagged in other regions (4.6 ± 10.8 d) ($p < 0.001$, Figs. 5A, 5C). The duration of each individual visit into the Cabrillo Pier array was also significantly longer for fish tagged in the vicinity of the pier (11.9 ± 5.2 hrs) than for fish tagged in other regions of the harbor (0.5 ± 1.7 hrs) ($p < 0.001$, Figs. 5B, 5D). California halibut spent an average of 95.9 ± 104.8 total days in the Cabrillo Pier area, with individual duration to the Pier area averaging 6.6 ± 5.3 hrs (Figs. 5E, 5F).

Table 2. Proportion of unique white croaker (WC) and California halibut (CH) detected by regions. Columns indicate region where fish were originally tagged and released, rows indicate regions where fish were subsequently detected. NA equates to no fish tagged in that region.

Region detected	Cabrillo Pier		LAOH		Fish Harbor		LAIH		LBIH		LBOH		Pier J		PV Shelf	
	WC	CH	WC	CH	WC	CH	WC	CH	WC	CH	WC	CH	WC	CH	WC	CH
LAOH	0.62	0.69	1	0.5	0.31	NA	0.13	0.25	0.08	0	0.04	0.5	0	0	0	0
Fish Harbor	0.1	0.58	0.07	0.5	1	NA	0	0.25	0.08	0	0.04	0.5	0	0.33	0	0
Angel's Gate	0.17	0.35	0.3	0.5	0.24	NA	0.13	0	0.08	0	0.04	0.5	0.08	0.17	0	0
Cabrillo Pier	0.97	1	0.74	0	0.17	NA	0.13	0	0.12	0	0	0	0	0.17	0	0
Queen's Gate	0.1	0.08	0.07	0.5	0	NA	0.04	0	0.16	0	0.04	0.5	0.16	0	0	0
PV Gate	0.03	0	0.11	0	0.1	NA	0.13	0.75	0.04	0	0.08	0	0	0	0	0
LBOH	0	0	0	0	0.07	NA	0.04	0.25	0.56	0.5	0.12	0.5	0	0.33	0	0
Pier J	0	0	0	0	0.03	NA	0	0.25	0.08	1	0	0	1	0	0	0
LAIH	0	0	0.15	0	0.03	NA	1	0	0.32	0	0	0	0.04	1	0	0
LBIH	0	0	0	0	0.03	NA	0.38	0	1	1	0.08	1	0	0	0	0

Table 3. LA-LB Harbors regional pairwise comparison of white croaker daily receiver presence from an ANOSIM test. CAB is short for Cabrillo Pier and Fish is short for Fish Harbor.

Regional comparison	Global R	P-value
LAOH v. Fish	0.883	0.001
LAOH v. CAB	0.646	0.001
LAOH v. LAIH	0.96	0.001
LAOH v. LBOH	0.944	0.001
LAOH v. LAIH	0.975	0.001
LAOH v. Pier J	0.998	0.001
Fish v. CAB	0.925	0.001
Fish v. LAIH	0.987	0.001
Fish v. LBOH	0.867	0.001
Fish v. LBIH	0.956	0.001
Fish v. Pier J	0.996	0.001
CAB v. LAIH	0.993	0.001
CAB v. LBOH	0.971	0.001
CAB v. LBIH	0.992	0.001
CAB v. Pier J	0.999	0.001
LAIH v. LBOH	0.914	0.001
LAIH v. LBIH	0.829	0.001
LAIH v. Pier J	1	0.001
LBOH v. LBIH	0.69	0.001
LBOH v. Pier J	0.975	0.001
LBIH v. Pier J	0.986	0.001

Discussion and Conclusions

Foraging strategies and habitat preferences likely influence the association of white croaker and California halibut to Cabrillo Pier. Both California halibut and white croaker showed an association to Cabrillo Pier; however, California halibut were in closer association to Cabrillo Pier and the ecotone of the low-lying reef. Most likely, white croaker were associating with the deeper habitat at the base of shelf that extends away from the Pier. In addition, California halibut showed a higher mean residency time (~ 96 d) compared to white croaker (~ 32 d) to the Cabrillo Pier area. These high mean residency times and closer association to ecotone (pier and reef) indicate that California halibut select these ecotone environments, most likely for increased foraging success due to the known higher prey densities compared to areas further from structure or reef (Ambrose and Anderson 1990; Anderson et al. 1989; Barros et al. 2001; Haaker 1975). In the temperate marine environment some reef-associated fishes acquire up to 25% of their diet from ecotone-associated prey (Johnson et al. 1994). Previous active tracking studies of juvenile California halibut in southern California estuaries found similar ecotone selection, where 54% of halibut position locations were within 2 m of eelgrass (*Zostera pacifica*) beds (Freedman et al. 2016). It is unclear if California halibut are associating to the ecotone provided by Cabrillo Pier or the sandy slope running parallel to the pier; however, it is clear they select for ecotone habitat, which puts them in closer proximity to Cabrillo Pier, resulting in a higher likelihood of being caught by fishers from the pier.

White croaker are commonly caught by anglers from fishing piers, including Cabrillo Pier, and were expected to have VPS rendered positions in close proximity to the Pier; however, the position data suggest that white croaker are not closely associating (<30 m) to Cabrillo Pier, but are selecting for the area slightly farther away (50-120 m). The sed-

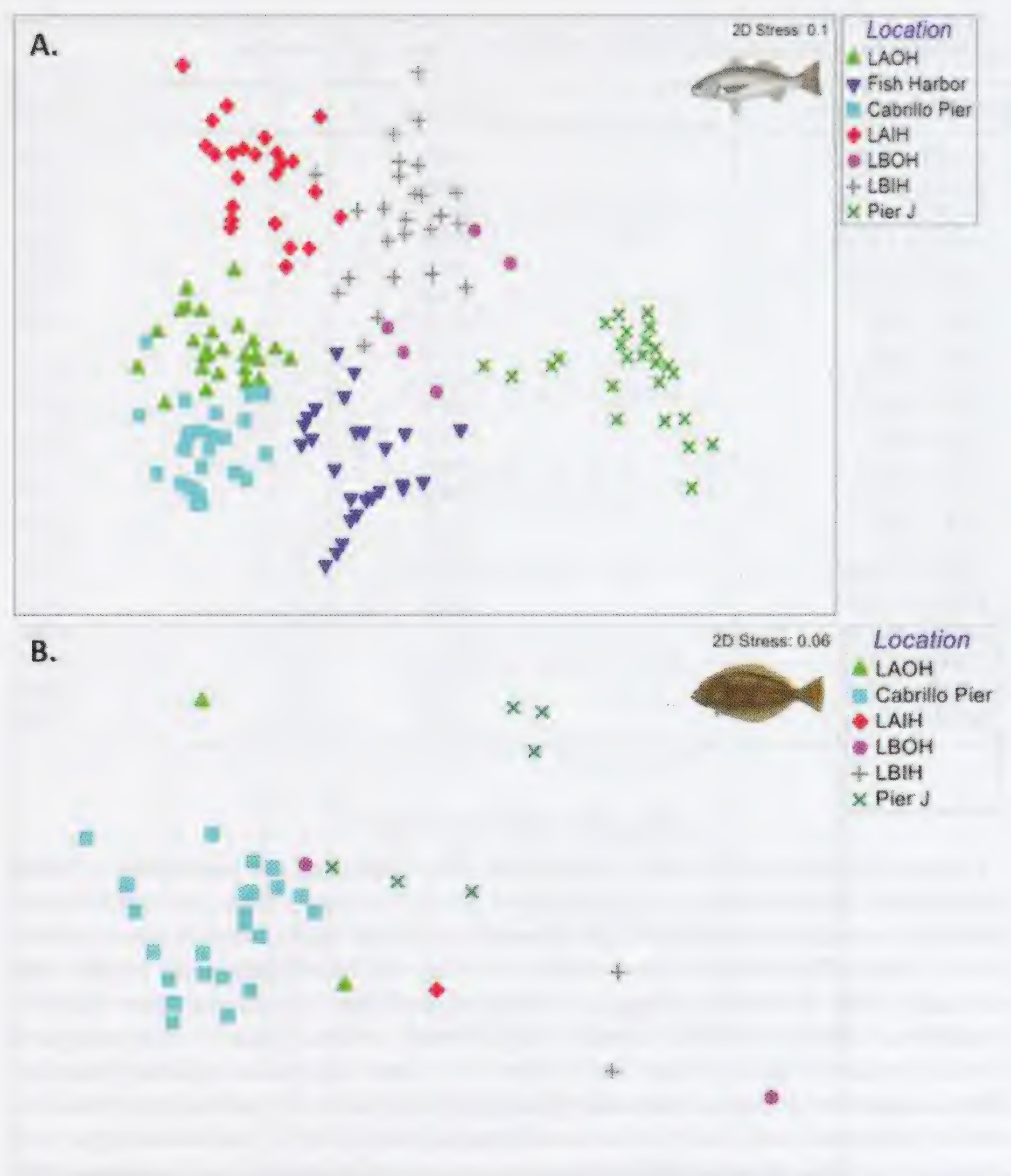


Fig. 2. Non-metric multidimensional scaling plot of receiver overlap of white croaker (A) and California halibut (B) from different tagging regions with the LA-LB Harbors after the dispersion weighting transformation. Tagging regions are as such: green triangles are LAOH, blue triangles are Fish Harbor, light blue squares are Cabrillo Pier, red diamonds are LAIH, pink circles are LBOH, grey crosses are LBIH, and dark green Xs are Pier J.

iment in the area 50-120 m from Cabrillo Pier consists of mud and silt, whereas in close proximity to Cabrillo Pier the sediment is primary comprised of sand (A. Barilotti, pers. obs.). Ahr et al. (2015) found that white croaker selected for areas of fine sediment grain size within the LA-LB Harbors, since it is correlated with higher sediment nutrient loads, high turbidity and increased densities of polychaete worms. This suggests that white croaker are not preferentially selecting areas in close proximity to Cabrillo Pier due to the lack of

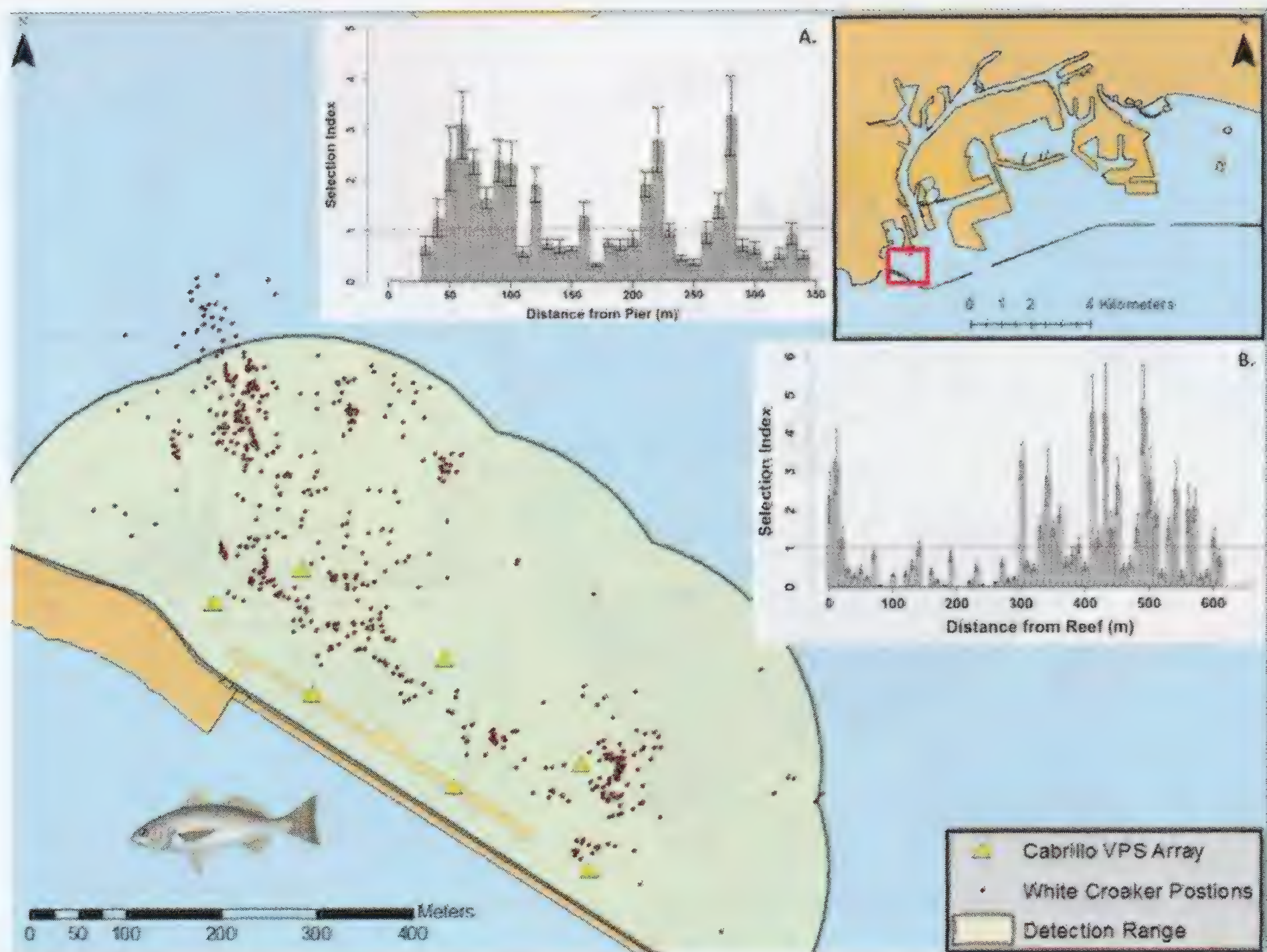


Fig. 3. Map of VPS-rendered positions of white croaker detected at Cabrillo Pier. Inset in upper right shows the Cabrillo Pier in the red box reference to the LA-LB Harbors. A.) Mean Pier selection index ($\pm$ SEM) of white croaker by distance (10 m bins) using VPS rendered positions. B.) Mean reef selection index ($\pm$ SEM) of white croaker by distance (10 m bins) using VPS rendered positions. Grey vertical line is the approximate distance of the bottom edge of the sand slope from Cabrillo Pier. The red line represents where fish randomly distributed would associate themselves; values above the red line equate to selection and value below the line indicate non-selection.

appropriate foraging habitat. Those individuals caught from Cabrillo Pier are most likely traveling nearer to the pier (50-100 m) and potentially lured towards the bait of fishers in an area they would not typically use.

Both white croaker and California halibut had virtually no VPS-rendered positions within 5 m of Cabrillo Pier, indicating neither species selects for close association for pier structure and the habitat directly under the pier. This was surprising since both species are caught frequently from Cabrillo Pier, and it was expected that some individuals would be detected underneath the pier. As an ambush predator, California halibut were expected to use shade provided by piers to strike at prey not in the shade (Able et al. 2013; Cermak 2002). One possibility why both species are avoiding the habitat directly associated with the pier is this habitat may have been too shallow as previous studies have found white croaker select for deeper areas within the LA-LB Harbors and coastal habitats (Ahr et al. 2015; Love et al. 1984). It is unlikely that California halibut are avoiding shallow water habitat as they are found in the surf-zone and in shallow estuaries (Haaker 1975), but their prey may be selecting deeper habitats to avoid aerial or surface predators such as plunge diving birds. However, another possibility for not detecting either species underneath or near the pier is that the pier pilings could be attenuating the acoustic signal and thereby reducing

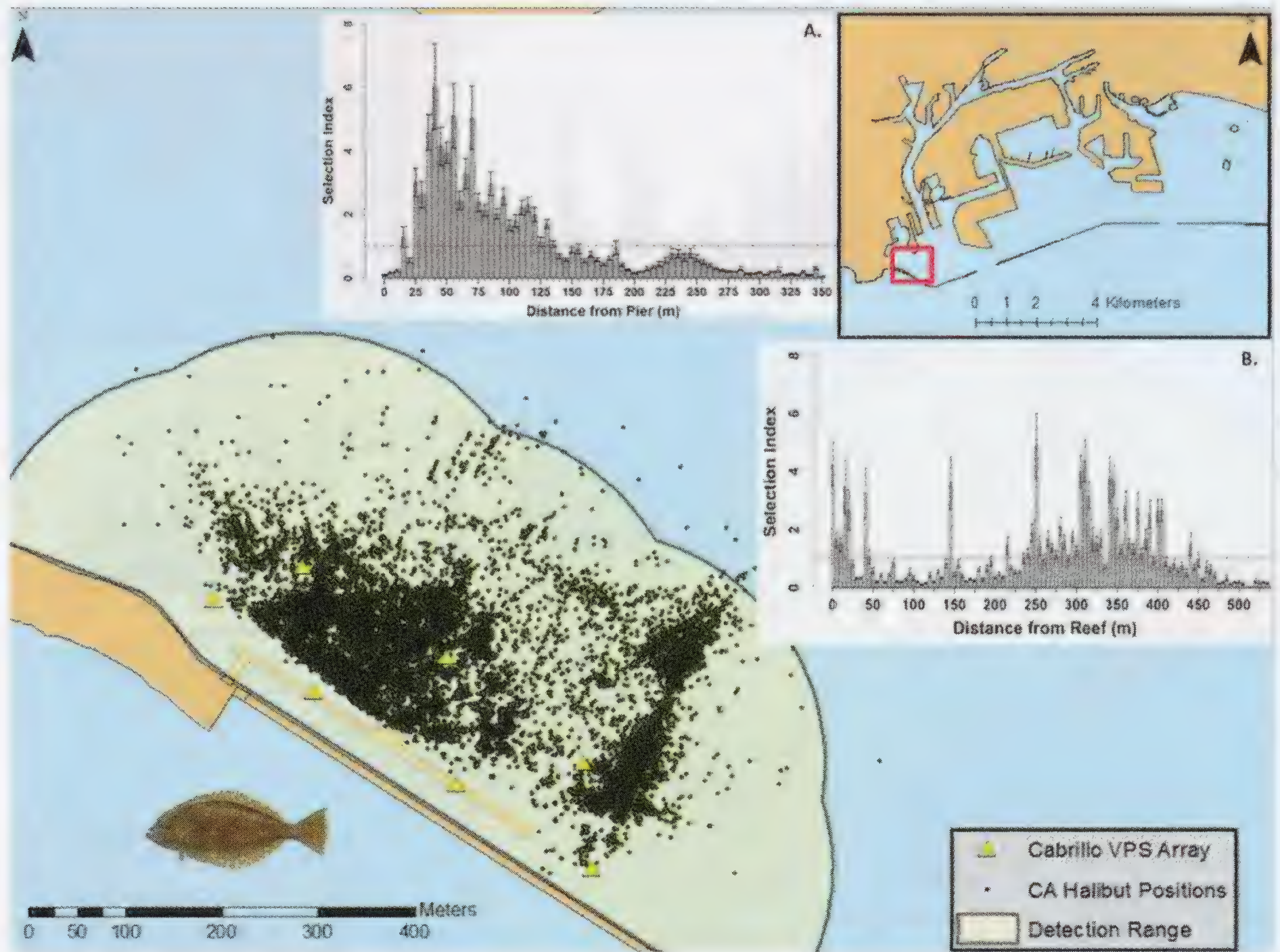


Fig. 4. Map of VPS-rendered positions of California halibut detected at Cabrillo Pier. Inset in upper right shows the Cabrillo Pier in the red box reference to the LA-LB Harbors. A.) Mean Pier selection index ($\pm$ SEM) of California halibut by distance (5 m bins) using VPS rendered positions. B.) Mean reef selection index ($\pm$ SEM) of California halibut by distance (5 m bins) using VPS rendered positions. Grey vertical line is the approximate distance of the bottom edge of the sand slope from Cabrillo Pier. The red line represents where fish randomly distributed would associate themselves; value above the red line equate to selection and value below the line indicate non-selection.

geo-position estimates for detections closer to the pier. Both receivers placed underneath Cabrillo Pier had reduced detection efficiency compared to the other five receivers in the Cabrillo Pier VPS array; however, it is unlikely all signals were blocked, since receivers in another study placed on offshore oil platform jackets did not experience large reductions in receiver performance (Anthony et al. 2012; Mireles et al. 2019). Overall, this indicates that both species are most likely not selecting this habitat, but anglers may be catching these species by fishing in the deeper water away from the pier.

Based on LA-LB Harbor-wide habitat characteristics, the area near Cabrillo Pier should provide suitable habitat for supporting white croaker (Ahr et al. 2015), and white croaker tagged at Cabrillo Pier were detected using the area for over a month. In addition, some white croaker with high levels of tissue contamination have been captured at Cabrillo Pier (Anderson et al. 2001), so it was expected to find individuals from other regions of the LA-LB Harbor using the areas around Cabrillo Pier. White croaker tagged in other areas of known sediment contamination within the LA-LB Harbors traveled to Cabrillo Pier; however, visitations were infrequent and those that were detected within the pier receiver array showed low affinity for the site. Previous movements studies have suggested that white croaker are nomadic (Farris et al. 2016; Wolfe and Lowe 2015), which is characterized by

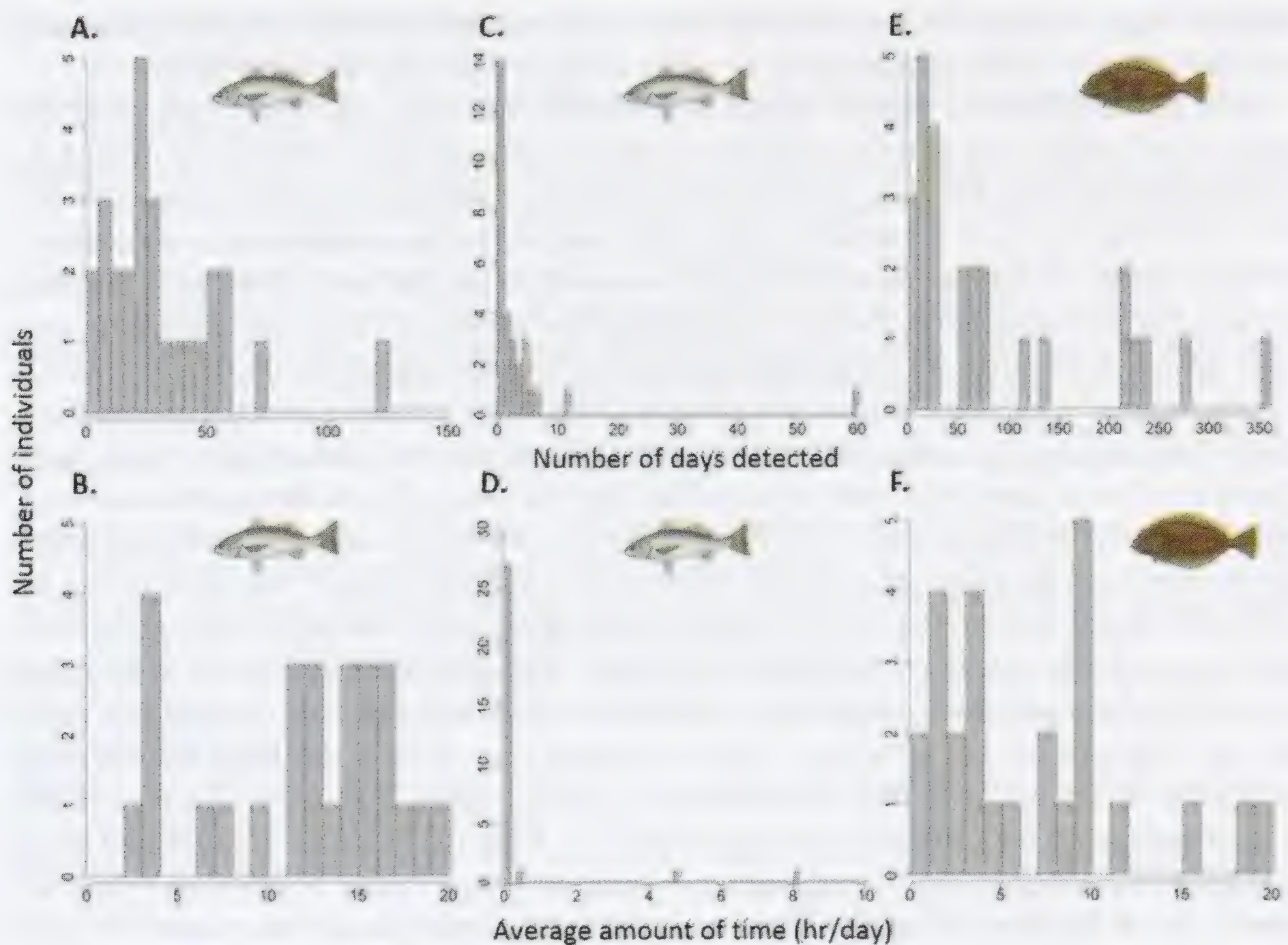


Fig. 5. The number of days individual white croaker tagged at Cabrillo Pier (A) and other regions (C) were detected within the Cabrillo Pier receiver array. The duration of time (hr/d) white croaker tagged at Cabrillo (B) and other regions (D) spent at the pier when detected. The number of days individual California halibut were detected at Cabrillo Pier (E) and the duration of time (hr/d) they spent at the pier when detected (F).

irregular movement behaviors that differ annually and are most likely the cause of inconsistent resource patches, periodic refuging from predators, or unsuitable environmental conditions (Berbert and Fagan 2012; Dean 1997; Lowe and Bray 2006). Thus, understanding and predicting white croaker movement patterns cannot only be a function of selecting for preferable habitat, but must incorporate individualistic patterns and unknown exogenous variables, which cause more movement variation. These unknown variables and nomadic behavior could explain why visiting white croaker moved on from Cabrillo Pier whereas those caught and tagged there remained for long periods of time.

Though many of the white croaker tagged from areas of higher known sediment contamination did not visit or stay at Cabrillo Pier for appreciable periods of time, white croaker found with the higher levels of tissue contamination are likely coming from these contaminated areas. Fish Harbor is the closest contaminated site to Cabrillo Pier and white croaker tagged in Fish Harbor had the highest probability of moving to Cabrillo Pier with five individuals visiting Cabrillo Pier and one staying for more than an hour. Though further away from Cabrillo Pier, LAIH has the highest levels of contaminants within the LA-LB Harbors (Anderson et al. 2001). White croaker tagged within LAIH displayed the highest degree of site fidelity of any area within the LA-LB Harbors, with some individuals detected for > 5 mos (Farris et al. 2016). However, when individuals did leave LAIH, they were found to move to all regions of the LA-LB Harbors (Farris et al. 2016). As a result,

white croaker in the LAIH have the potential to accumulate unsafe levels of contaminants and then disperse these contaminants to other areas as well as up the food chain.

No California halibut initially tagged in a known contaminated region were observed moving to Cabrillo Pier, and the greatest connectivity was observed between the outer LA-LB Harbor receivers. The lack of connectivity between contaminated regions and Cabrillo Pier could be a result of a relatively small inner LA-LB Harbor sample size ($n = 6$). However, no California halibut ($n = 36$) tagged in outer LA-LB Harbor regions were detected moving into the inner harbor regions, suggesting that inner LA-LB Harbors could potentially have few resources for supporting an adult California halibut population. A translocation study of California halibut in southern Californian estuaries concluded that California halibut select for areas with higher prey delivery potential, where individuals were found to relocate from backwater areas of low flow to areas of high flow within 24 hrs of translocation (Freedman et al. 2015; Freedman et al. 2016). If California halibut do select for areas of high flow and greater prey delivery, then it would explain why none of the individuals tagged in the outer LA-LB Harbors were detected by the inner harbor receivers, and why half those tagged in the inner regions left. Those individuals caught in the inner LA-LB Harbors potentially recruited to the area as larval fish and used the area as a nursery. As California halibut grow they experience ontogenetic shifts in diet from more benthic associated invertebrates to more epibenthic and pelagic fishes. To sustain this diet of fish it is advantageous for California halibut to emigrate from the inner harbor and find areas that have increased prey delivery, resulting in a relatively small adult population within the inner LA-LB Harbors. Thus, the likelihood of catching individuals with potentially high levels of tissue contamination emigrating from the inner LA-LB Harbor regions may be relatively infrequent.

This is the first known study to examine the influence of California fishing piers on the association and movements of fishes. This study only examined one pier within a large harbor complex where other piers are available; pier habitat may be important to fish in areas where there is less vertical structure available or in non-protected habitats (e.g. piers on surf exposed beaches). Also, to understand the effectiveness of pier structure as habitat for fish, future research should use fish species that are known to associate with reef structure (e.g. *Paralabrax* spp., *Sebastes* spp.), since they should show a higher affinity for the structure provided by piers. Piers, both public and commercial, are integral parts of the waterfront, and understanding their influence as fish habitat should be included in future fish research.

Acknowledgments

The authors would like to thank the numerous volunteers from CSULB and the Shark Lab including Jillian Sawyna, Ryan Logan, Sarah Luongo, Garry Teesdale, Yvette Ralph, and Angela Lyons. John Ballotti and the anglers of the The Los Angeles Rod and Reel Club for providing fishing support. We would also like to thank Southern California Marine Institute for supporting field operations for this project. Funding was provided by the Port of Los Angeles, Port of Long Beach, and SCTC Marine Biology Foundation.

Literature Cited

- Able, K.W., Grothues, T.M., and Kemp, I.M. 2013. Fine-scale distribution of pelagic fishes relative to a large urban pier. *MEPS*, 476:185-198.
- Ahr, B., Farris, M., and Lowe, C.G. 2015. Habitat selection and utilization of white croaker (*Genyonemus lineatus*) in the Los Angeles and Long Beach Harbors and the development of predictive habitat use models. *Mar. Environ. Res.*, 108:1-13.

- Alevras, R.A., and Edwards, S.J. 1985. Use of reef-like structures to mitigate habitat loss in an estuarine environment. *Bull. Mar. Sci.*, 37:396.
- Ambrose, R.F., and Anderson, T.W. 1990. Influence of an artificial reef on the surrounding infaunal community. *Mar. Biol.*, 107:41-52.
- Anderson, B.S., Hunt, J.W., Phillips, B.M., Fairey, R., Roberts, C.A., Oakden, J.M., Puckett, H.M., Stephenson, M., Tjeerdema, R.S., and Long, E.R. 2001. Sediment quality in Los Angeles Harbor, USA: A triad assessment. *Environ. Toxicol. Chem.*, 20:359-370.
- Anderson, T.W., DeMartini, E.E., and Roberts, D.A. 1989. The relationship between habitat structure, body size and distribution of fishes at a temperate artificial reef. *Bull. Mar. Sci.*, 44:681-697.
- Anthony, K.M., Love, M.S., and Lowe, C.G. 2012. Translocation, homing behavior and habitat use of groundfishes associated with oil platforms in the East Santa Barbara Channel, California. *BSCAS*, 111:101-118.
- Baine, M. 2001. Artificial reefs: A review of their design, application, management and performance. *Ocean Coast Manag.*, 44:241-259.
- Barros, F., Underwood, A.J., and Lindegarth, M. 2001. The influence of rocky reefs on structure of benthic macrofauna in nearby soft-sediments. *Estuar. Coast. Shelf Sci.*, 52:191-199.
- Berbert, J.M., and Fagan, W.F. 2012. How the interplay between individual spatial memory and landscape persistence can generate population distribution patterns. *Ecol. Complex.*, 12:1-12.
- Blasius, M.E., and Goodmanlowe, G.D. 2008. Contaminants still high in top-level carnivores in the Southern California Bight: Levels of DDT and PCBs in resident and transient pinnipeds. *Mar. Pollut. Bull.*, 56:1973-1982.
- Bohnsack, J.A. 1989. Are high densities of fishes at artificial reefs the result of habitat limitation or behavioral preference? *Bull. Mar. Sci.*, 44:631-645.
- Brown, D.W., McCain, B.B., Horness, B.H., Sloan, C.A., Tilbury, K.L., Pierce, S.M., Burrows, D.G., Chan, S.L., Landahl, J.T., and Krahn, M.M. 1998. Status, correlations and temporal trends of chemical contaminants in fish and sediment from selected sites on the Pacific coast of the USA. *Mar. Pollut. Bull.*, 37:67-85.
- Butler, A.J., and Connolly, R.M. 1999. Assemblages of sessile marine invertebrates: still changing after all these years? *MEPS*, 182:109-118.
- Carr, M.H. 1991. Habitat selection and recruitment of an assemblage of temperate zone reef fishes. *J. Exp. Mar. Biol. Ecol.*, 146:113-137.
- Caselle, J.E., Love, M.S., Fusaro, C., and Schroeder, D. 2002. Trash or habitat? Fish assemblages on offshore oilfield seafloor debris in the Santa Barbara Channel, California. *ICES J. Mar. Sci.*, 59:258-265.
- Cermak, M.J. 2002. *Caranx latus* (*Carangidae*) chooses dock pilings to attack silverside schools: A tactic to interfere with stereotyped escape behavior of prey? *Biol. Bull.*, 203:241-243.
- Clarke, K.R.G., Somerfield, P.J., Warwick, R.M. 2014. Change in marine communities: an approach to statistical analysis and interpretation. *PRIMER-E Ltd.*, 9-5 pp.
- Conner, L.M., and Plowman, B.W. 2001. Using Euclidean distances to assess nonrandom habitat use. *Academic Press*, 275-290 pp.
- Dean, W.R.J. 1997. The distribution and biology of nomadic birds in the Karoo, South Africa. *J. Biogeogr.*, 24:769-779.
- Deysher, L.E., Dean, T.A., Grove, R.S., and Jahn, A. 2002. Design considerations for an artificial reef to grow giant kelp (*Macrocystis pyrifera*) in Southern California. *ICES J. Mar. Sci.*, 59:201-207.
- Dorn, P., Johnson, L., and Darby, C. 1979. The swimming performance of nine species of common California inshore fishes. *T. Am. Fish. Soc.*, 108:366-372.
- Eganhouse, R.P., and Pontolillo, J. 2008. DDE in sediments of the Palos Verdes Shelf, California: In situ transformation rates and geochemical fate. *Environ. Sci. Technol.*, 42:6392-6398.
- Espinoza, M., Farrugia, T.J., Webber, D.M., Smith, F., and Lowe, C.G. 2011. Testing a new acoustic telemetry technique to quantify long-term, fine-scale movements of aquatic animals. *Fish. Res.*, 108:364-371.
- Farris, M., Ahr, B., and Lowe, C.G. 2016. Area use and movements of the white croaker (*Genyonemus lineatus*) in the Los Angeles and Long Beach Harbors. *Mar. Environ. Res.*, 120:145-153.
- Freedman, R., Whitcraft, C.R., and Lowe, C.G. 2015. Connectivity and movements of juvenile predatory fishes between discrete restored estuaries in southern California. *MEPS*, 520:191-201.
- Freedman, R.M., Espasandin, C., Holcombe, E.F., Whitcraft, C.R., Allen, B.J., Witting, D., and Lowe, C.G. 2016. Using movements and habitat utilization as a functional metric of restoration for estuarine juvenile fish habitat. *Mar. Coast. Fish.*, 8:361-373.

- Garcia, J., Mourier, J., and Lenfant, P. 2015. Spatial behavior of two coral reef fishes within a Caribbean marine protected area. *Mar. Environ. Res.* 109:41-51.
- Haaker, P.L. 1975. The biology of the California halibut, *Paralichthys californicus* (Ayres) in Anaheim Bay. Lane and CW Hill (eds.). California Department of Fish and Game Fish Bulletin. 165:137-159.
- Johnson, T.D., Barnett, A.M., DeMartini, E.E., Craft, L.L., Ambrose, R.F., and Purcell, L.J. 1994. Fish production and habitat utilization on a southern California artificial reef. *Bull. Mar. Sci.*, 55:709-723.
- Jones, K. 2004. Pier fishing in California. Publishers Design Group.
- Kellison, T.G., and Sedberry, G.R. 1998. The effects of artificial reef vertical profile and hole diameter on fishes off South Carolina. *Bull. Mar. Sci.*, 62:763-780.
- Keough, M.J. 1983. Patterns of recruitment of sessile invertebrates in two subtidal habitats. *J. Exp. Mar. Biol. Ecol.*, 66:213-245.
- Longnecker, M.P., Rogan, W.J., and Lucier, G. 1997. The human health effects of DDT (Dichlorodiphenyl-trichloroethane) and PCBs Polychlorinated Biphenyls and an overview of organochlorines in public health. *Annu. Rev. Public Health*, 18:211-244.
- Love, M.S., McGowen, G.E., Westphal, W., Lavenberg, R.J., and Martin, L. 1984. Aspects of the life history and fishery of the white croaker, *Genyonemus lineatus* (Sciaenidae), off California. *Fish. Bull.*, 82:179-198.
- Lowe, C.G., and Bray, R.N. 2006. Movement and activity patterns. Pp. 524-553 in *The Ecology of Marine Fishes: California and Adjacent Waters*. (L.G. Allen and M.H. Horn, ed.) Univ. of California Press.
- Lowe, C.G., Topping, D.T., Cartamil, D.P., and Papastamatiou, Y.P. 2003. Movement patterns, home range, and habitat utilization of adult kelp bass *Paralabrax clathratus* in a temperate no-take marine reserve. *MEPS*, 256:205-216.
- Malins, D.C., McCain, B.B., Brown, D.W., Myers, M.S., Krahn, M.M., and Chan, S.L. 1987. Toxic chemicals, including aromatic and chlorinated hydrocarbons and their derivatives, and liver lesions in white croaker (*Genyonemus lineatus*) from the vicinity of Los Angeles. *Environ. Sci. Technol.*, 21:765-770.
- Martin, C.J.B., and Lowe, C.G. 2010. Assemblage structure of fish at offshore petroleum platforms on the San Pedro Shelf of southern California. *Mar. Coast. Fish.*, 2:180-194.
- Meyer, C.G., Papastamatiou, Y.P., and Clark, T.B. 2010. Differential movement patterns and site fidelity among trophic groups of reef fishes in a Hawaiian marine protected area. *Mar. Biol.*, 157:1499-1511.
- Mireles, C., Martin, C.J.B., and Lowe, C.G. 2019. Site fidelity, vertical movement, and habitat use of nearshore reef fish on offshore petroleum platforms in southern California. *Bull. Mar. Sci.*, 95:657-682.
- Mull, C.G., Lyons, K., Blasius, M.E., Winkler, C., O'Sullivan, J.B., and Lowe, C.G. 2013. Evidence of maternal offloading of organic contaminants in white sharks (*Carcharodon carcharias*). *PLoS One*, 8(4): 8.
- Papastamatiou, Y., Meyer, C.G., Kosaki, R.K., Wallsgrove, N.J., and Popp, B.N. 2015. Movements and foraging of predators associated with mesophotic coral reefs and their potential for linking ecological habitats. *MEPS*, 521:155-170.
- Rilov, G., and Benayahu, Y. 1998. Vertical artificial structures as an alternative habitat for coral reef fishes in disturbed environments. *Mar. Environ. Res.*, 45:431-451.
- Schroeter, S.C., Reed, D.C., and Raimondi, P.T. 2015. Effects of reef physical structure on development of benthic reef community: A large-scale artificial reef experiment. *MEPS*, 540:43-55.
- Teesdale, G.N., Wolfe, B.W., and Lowe, C.G. 2015. Patterns of home ranging, site fidelity, and seasonal spawning migration of barred sand bass caught within the Palos Verdes Shelf Superfund Site. *MEPS*, 539:255-269.
- Wolfe, B.W., and Lowe, C.G. 2015. Movement patterns, habitat use and site fidelity of the white croaker (*Genyonemus lineatus*) in the Palos Verdes Superfund Site, Los Angeles, California. *Mar. Environ. Res.*, 109:69-80.
- Zeng, E.Y., and Venkatesan, M.I. 1999. Dispersion of sediment DDTs in the coastal ocean off southern California. *Sci. Total. Environ.*, 229:195-208.

Protecting the Wildland-Urban Interface in California: Greenbelts vs Thinning for Wildfire Threats to Homes

Jon E. Keeley,^{1,2*} Greg Rubin,³ Teresa Brennan,¹ and Bernadette Piffard³

¹*U.S. Geological Survey, Western Ecological Research Center, Sequoia-Kings Canyon Field Station, 47050 General's Highway, Three Rivers, CA 93271*

²*Department of Ecology and Evolutionary Biology, University of California, 612 Charles E. Young Drive, South Los Angeles, CA 90095*

³*California's Own Native Landscape Design, Inc., 25950 Los Arboles Ranch Rd., Escondido, CA 92026*

Abstract.—This study utilized native chaparral and sage scrub shrubs planted in lightly irrigated greenbelts around homes to evaluate the impact on live fuel moisture content (LFMC) and predicted fire behavior. As to be expected LFMC varied markedly throughout the year being over 100% in winter in all species and treatments that included adjacent thinned native shrublands and untreated control shrublands. However, in the summer and fall there were marked differences between treatments. For most species lightly irrigated plants had the highest LFMC in the summer and fall, followed by thinned treatments and controls. These differences in moisture content coupled with structural differences in the vegetation contributed to expected differences in flame length and rate of spread. Lightly irrigated native shrubs planted around homes can reduce fire hazard while possessing other desirable features of utilizing native vegetation.

Losses of lives and property from wildfires have spiraled out of control in recent years and we need to address potential strategies for reducing the impact of fires that spread into urban communities (Keeley and Syphard 2019). Key to saving homes and lives is the 'defensible space' around homes (Syphard et al. 2014; Penman et al. 2019). Ordinances are largely focused on reducing wildland fuels adjacent to homes. However, widespread clearing of native vegetation near homes (Fig. 1) impacts aesthetics, privacy, biodiversity, faunal attractiveness, and property values (Gibbons et al. 2018).

Hazard reduction strategies involving options other than clearance, such as creation of 'greenbelts' (Gardner et al. 1987; Kent 2005), are solutions not yet fully explored; greenbelts of vegetation (i.e., widely space irrigated shrubs and trees) were applied in numerous sites and seemed to be effective at reducing fire hazard. Other studies have shown that the 'greenness' (measured by the Normalized Vegetation Difference Index) near homes had the same effect as removing woody vegetation (Gibbons et al. 2018). This is effective primarily because fire behavior is markedly influenced by live fuel moisture content (LFMC), a measure of the water content of vegetation (Dennison et al. 2008). For chaparral shrublands of southern California, when LFMC of shrubs is 60 - 80%, fire risk is considered to be high because the heat energy generated by a burning plant exceeds the energy needed to eliminate plant moisture, leaving excess energy to pre-heat adjacent plants and propagate a wildfire (Weise et al. 1998, Stow et al. 2005, Dennison et al. 2010). There is empirical

* Corresponding author: jon_keeley@usgs.gov



Fig. 1. Example of an increasingly common type of fuel clearance around homes near Ramona in San Diego County (Photo Jon Keeley, USGS).

evidence that LFMC will affect rate of fire spread in both laboratory and field studies (Rossa and Fernandes 2018).

Greenbelts have often been favored because esthetically they are more pleasing than heavy clearance that removes much of the vegetation in belts around properties (e.g., Fig. 1). One of the downsides of greenbelts as often applied is that they commonly involve non-native species, and homeowners often move to the wildland-urban interface in order to experience natural settings. An alternative is to use native shrubs in landscapes around homes and maintain them with light watering during the summer. This allows yards with native flora but kept lightly irrigated during the fire season to potentially inhibit fire spread. This approach has been utilized in a number of landscaping projects in southern California (Rubin and Warren 2013). It has advantages over many standard greenbelt designs in that it uses native shrubs, which are most attractive to native fauna (Jeschke et al. 2014), adding to the wildland-urban experience.

While there is anecdotal evidence of the effectiveness of such light irrigation of native plantings at reducing fire hazard (e.g., Pellizzoro et al. 2007), there is little in the way of experimental evidence. The purpose of this project was to test the efficacy of this technique at altering fire behavior in comparison with more commonly used shrub ‘clearance’ or ‘thinning’ or with untreated native shrublands.

Materials and Methods

We utilized three home landscaping projects installed by California’s Own Native Landscape Design Inc. (Escondido, CA) that had been installed at least 5 yrs prior to the experiment comprising shrub species native to the site and lightly irrigated (see below) throughout the dry season. Just beyond the property line were native shrublands that had been thinned in the years immediately before our experiment. Just beyond the thinned vegetation was untreated native shrublands. Based largely on differences in live fuel moisture

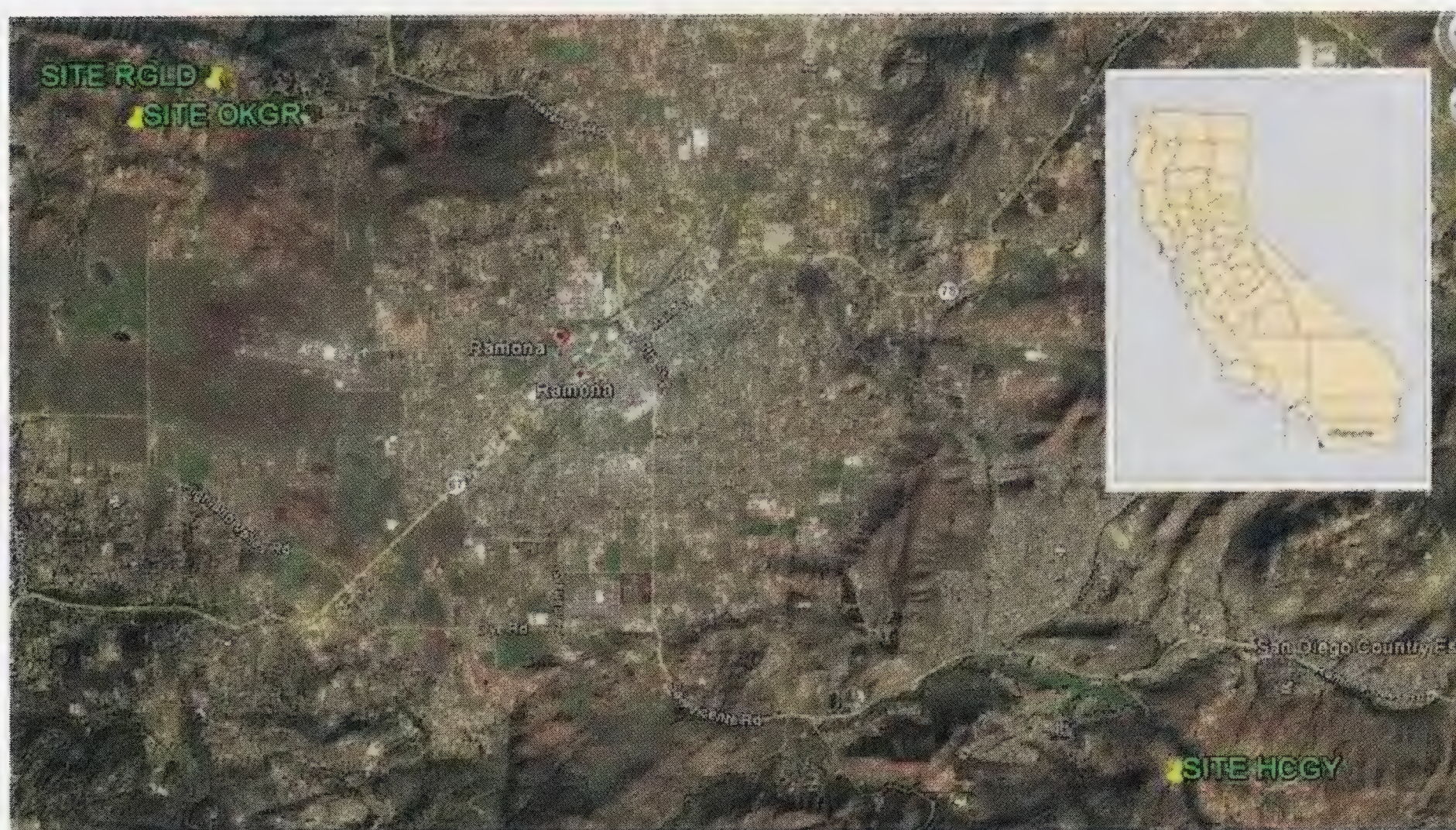


Fig. 2. Aerial view of Ramona showing location of the three residences used for this study (Google Earth™ image). These sites were on High Country Rd. (HCGY), Oak Grove Rd. (OKGR), and Rangland Rd. (RGLD) in the community of Ramona, San Diego County, CA.

during the fire season we hypothesized that the lightly irrigated treatments would generate lower flame lengths and slower rates of fire spread than either unirrigated thinned treatments or untreated controls.

Our experimental design was to monitor live foliage fuel moisture approximately every 2 wks for 2.5 yrs on irrigated, thinned and untreated chaparral and sage scrub species at three sites. In addition, we quantified the extent of thinning by measuring areal coverage of thinned and untreated vegetation. Using these data, we modeled fire behavior with Fuel and Fire Tools software¹ for the two treatments and untreated controls. These modeling results were used to evaluate our hypothesis.

The overall experimental design was to compare the predicted fire behavior in three locations in San Diego county's wildland-urban interface in order to evaluate the effectiveness of different approaches in creating defensible space at the WUI. The criteria for site selection was a wildland-urban interface that included a residence adjacent to wildland open space. Three homes in northeastern San Diego County near Ramona California were selected for study (see Fig. 2).

At each site, three study areas were established surrounding the house to include 1) native plants indigenous to the site were planted in an area 10 m from the home, 5-10 m wide and at least 30 m long (e.g., Fig. 3), 2) at least 30 m from the home a native natural sage scrub or chaparral shrubland area that had been thinned according to prescriptions commonly used for creating defensible space around homes, and 3) further from the home a control area of natural shrubland vegetation. All sites were more or less on level ground with slopes < 10%.

¹ <https://www.fs.fed.us/pnw/fera/fft/index.shtml>



Fig. 3. Example of summer lightly irrigated native vegetation (photo by Jon Keeley, USGS).

The prescription for the lightly irrigated plantings was: Native shrubs indigenous to the site were planted roughly 10 m from the homes in a belt of approximately 5 m width surrounding the home extending 30-50 m. These were lightly irrigated during the summer months June to October, once every 10-14 days with a MP-RotatorTM stream nozzles with rate of 9.75 mm equivalent precipitation per hour; this equated to 12.5 to 18.8 mm per month during the summer.

The native shrubland vegetation outside the lightly irrigated vegetation were areas dominated by lower stature sage scrub and others with higher stature chaparral species. These areas were matched in the thinned and untreated areas and comprised the same sage scrub species, *Eriogonum fasciculatum* (California buckwheat) and dependent on the site the same chaparral component; *Arctostaphylos glauca*, *Quercus berberidifolia* and *Malosma laurina*.

Data were collected on fuel structure within the three locations in order to select the appropriate fuel model for the fire behavior part of this study. Species composition varied between the three sites and although this adds to the variance in data, it provides a better picture of a range of responses in different settings. In general, each site comprised species characteristic of both sage scrub and chaparral. Structural characteristics were determined with a single 20 m line transect through each treatment in each vegetation type at each site, recording area of live and dead foliage and height for each species using the line-intercept method (Cox 1990).

Species were selected for live fuel moisture content based on availability. At all three sites we sampled the semi-deciduous sage scrub *Eriogonum fasciculatum*. An evergreen chaparral shrub was also included but varied between sites: *Arctostaphylos glauca* at Site HCGY, *Malosma laurina* at Site OKGR, and *Quercus berberidifolia* at Site RGLD (See Fig. 2).

LFMC was determined approximately every 2 wks for 2 yrs according to established procedures (Countryman and Dean 1979; Pollet and Brown 2007; Haase et al. 2016). Briefly, three terminal foliage samples were collected from each of the target species in each

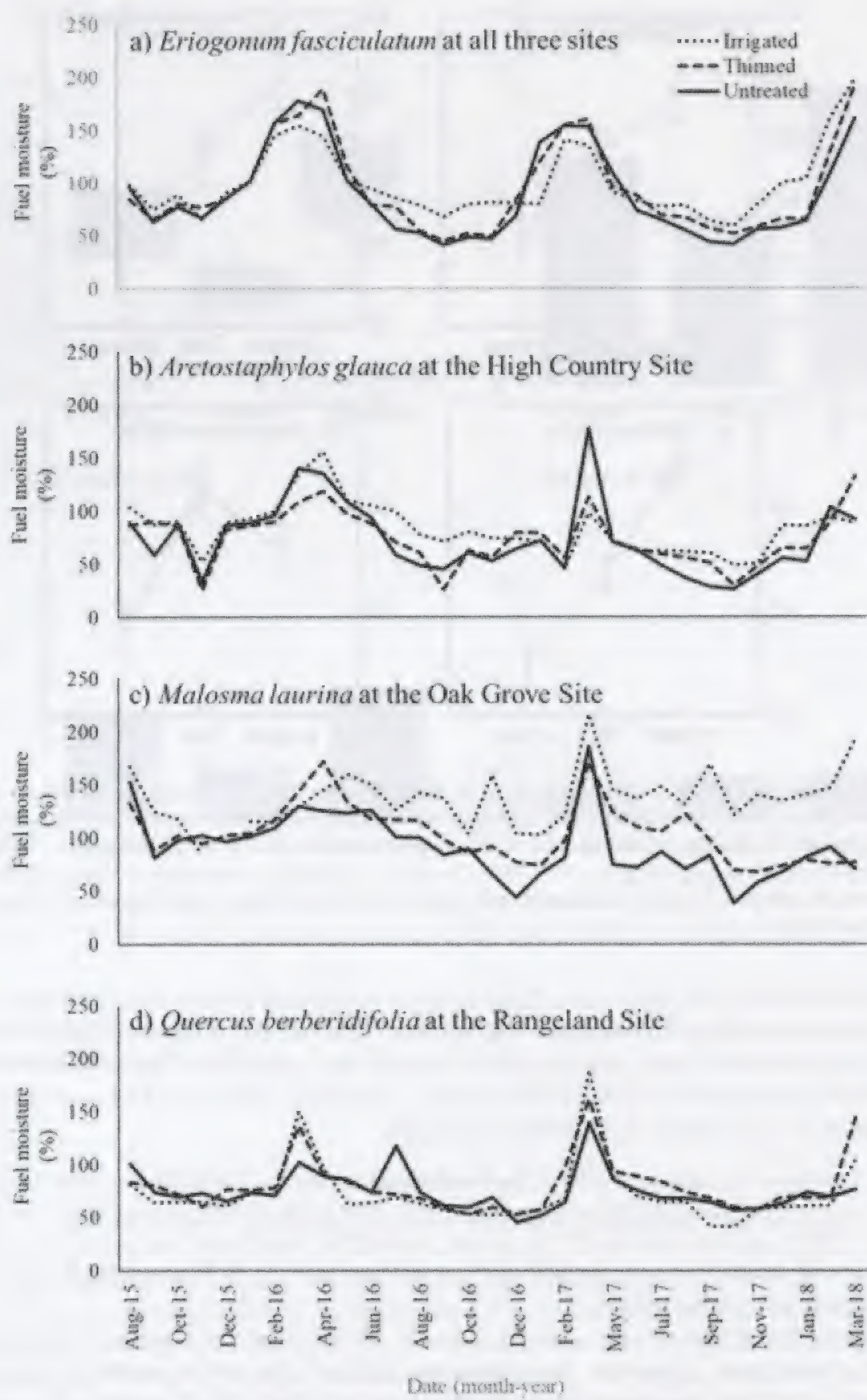


Fig. 4. Two and a half years of live fuel moisture content (LFMC) for the four species used in this study presented by treatment.

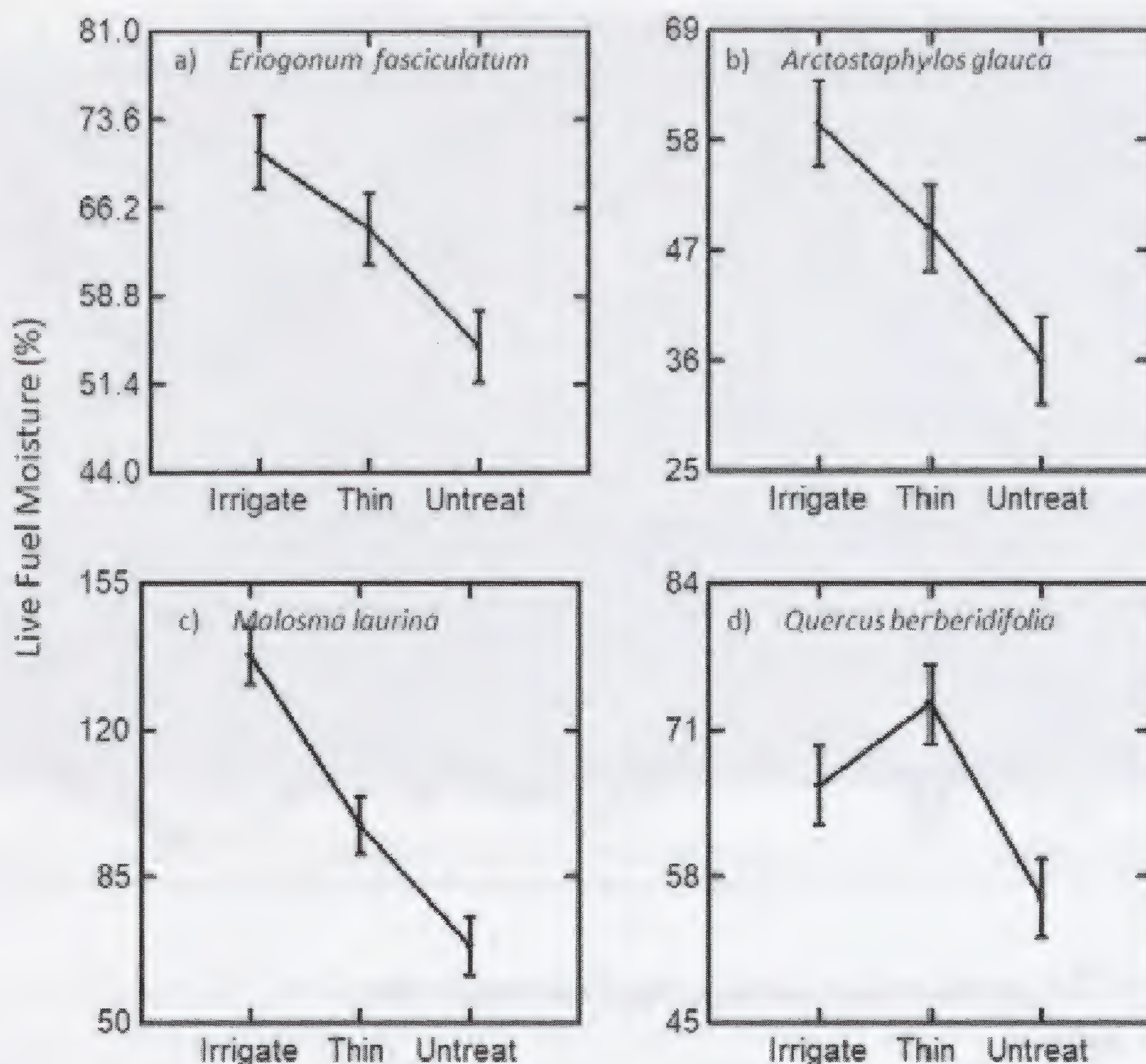


Fig. 5. Average LFMFC for all collections from June - October 2017 for the four species. Error bars are the standard error of the mean and ANOVA gave a significant difference between treatments for all species, though the Bonferroni test showed: a) *Eriogonum fasciculatum*: $P < 0.001$, Irrigated = Thinned > Untreated, b) *Arctostaphylos glauca*: $P = 0.004$, Irrigated > Thinned > Untreated, c) *Malosma laurina*: $P < 0.001$, Irrigated > Thinned > Untreated, and d) *Quercus berberidifolia*: $P = 0.04$, Irrigated = Thinned > Untreated.

treatment at each of the three sites. These samples were stored in pre-weighed airtight tins and returned to the lab and weighed. The lids were removed and dried to constant weight in a forced convection oven and reweighed. Samples were collected at the same time each day after morning dew had dissipated (between 11AM and 2 PM). Live fuel moisture was calculated as a percentage of dry weight, as follows:

$$\text{Live fuel moisture (\%)} = \frac{((\text{wet wt} - \text{tin weight}) - (\text{dry wt} - \text{tin wt}))}{(\text{dry wt} - \text{tin wt})} * 100$$

Summer and fall data were compared across treatments with ANOVA and the Bonferroni test for separating treatments.

Expected fire behavior was modeled for each of our treatments using the Fuel and Fire Tools software application that allows the input of site specific vegetation and fuels data as well as various environmental characteristics including fuel moisture, slope, and wind speed. The vegetation data from our treatments that we incorporated into the model (Appendix I) included the overall average height and percentage live and dead cover of vegetation in each treatment as well as the relative cover of each species present based on the line-transect data. We did not collect data on fuel moisture of downed woody fuels and

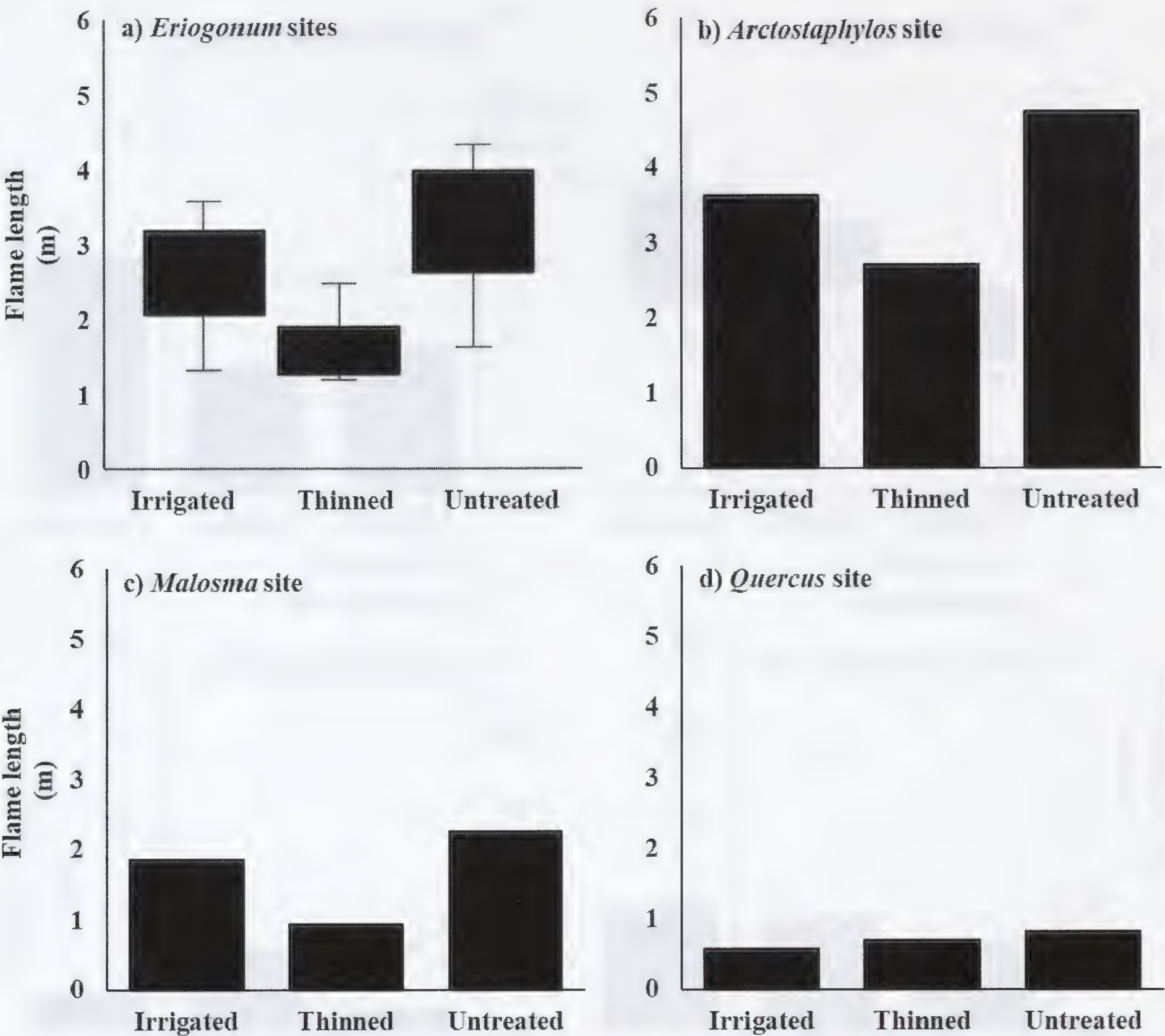


Fig. 6. Predicted flame length for the dry season for a) *Eriogonum*, error bars are the standard error of the mean across the three sites, b) *Arctostaphylos*, c) *Malosma*, and d) *Quercus* sites for the three treatments.

therefore accepted the default values associated with these specific vegetation types that are included in the model.

Results

As to be expected LFMC varied markedly throughout the year nearly always over 100% in winter in all species and treatments (Fig. 4). Summer and fall patterns were of particular interest as this is the time of most wildfires in the region. These warm season LFMC varied between treatments and species (Fig. 5). For the sage scrub species *Eriogonum fasciculatum* it was not significantly different between irrigated and thinned treatments but these were higher than untreated controls (Fig. 5a). Evergreen species had the highest LFMC in irrigated plants, followed by thinned and then controls for *Arctostaphylos glauca* and *Malosma laurina* (Fig. 5b, c). These species, however, differed significantly in LFMC, with irrigated *M. laurina* having over double that of irrigated *A. glauca*. *Quercus berberidifolia* showed no difference between irrigated and thinned shrubs but they were slightly higher than controls (Fig. 5d). In short, during the warm season untreated controls had lower LFMC than treated plants for all for target species. Only the two evergreen shrubs *A. glauca* and

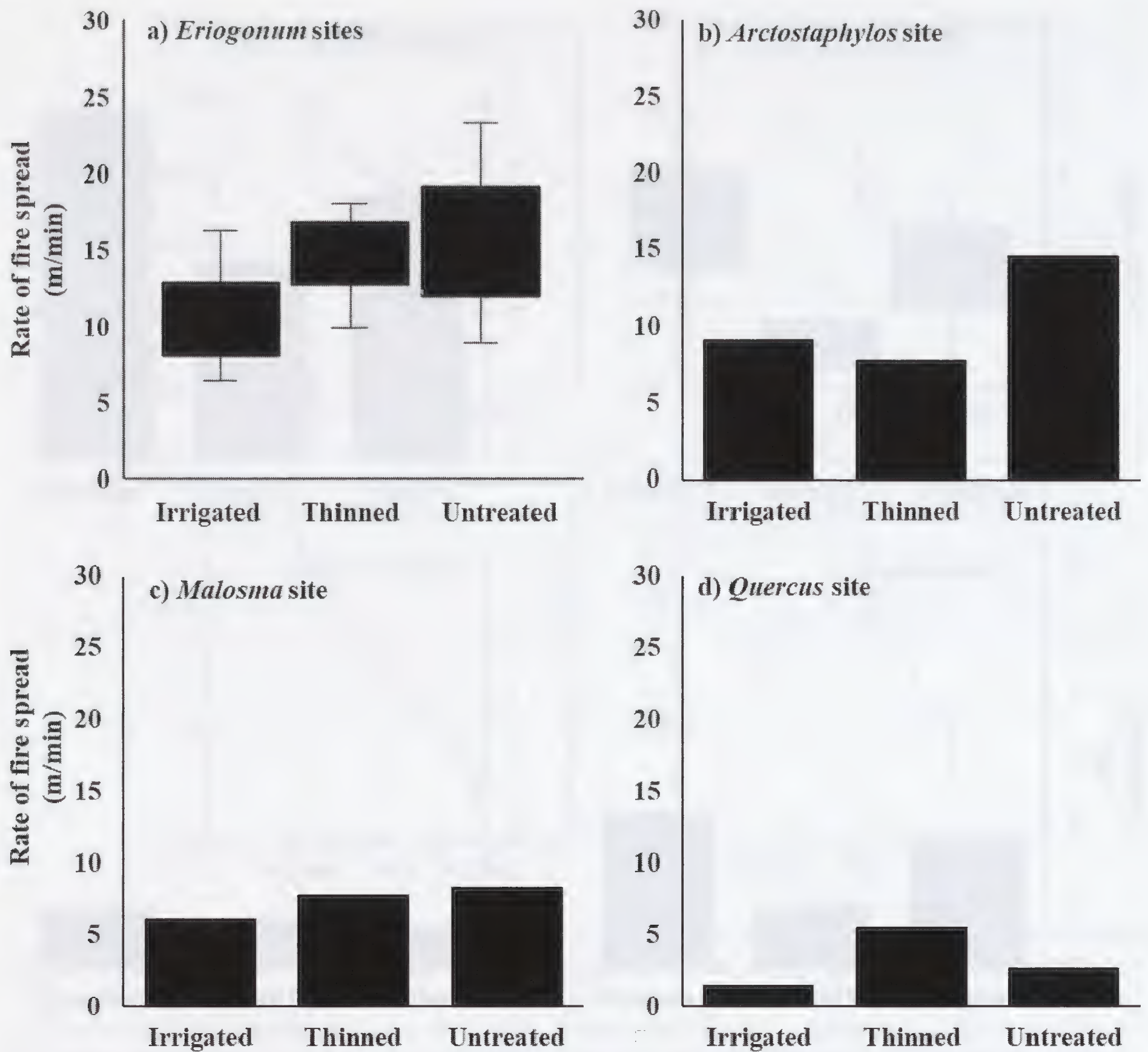


Fig. 7. Predicted rate of fire spread for the dry season for a) *Eriogonum*, error bars are the standard error of the mean across the three sites, b) *Arctostaphylos*, c) *Malosma*, and d) *Quercus* sites for the three treatments.

M. laurina had higher fuel moisture in irrigated over thinned plants; for *E. fasciculatum* and *Q. berberidifolia* there was no difference between irrigated and thinned plants.

At all sites chaparral was markedly taller than the sage scrub but within a site for chaparral and sage scrub, heights were roughly similar across treatments (Appendix I). Vegetative cover was similar between controls and irrigated plantings and markedly lower in thinned treatments.

Our fire behavior models were run for each site separately but the general patterns of treatments were similar for *Eriogonum fasciculatum* and these results were combined for all sites (Fig. 6a) and compared with the evergreen species at their single sites (Fig. 6b-d). Untreated controls would be expected to have higher flame lengths and this was true for all species (Fig. 6). However, thinned plots with their much lower biomass were predicted to have the lowest flame lengths. Irrigated treatments, however, seemed to stand out most in the lower rates of fire spread (Fig. 7). Fuel structure played an important role in that the fine fuels in *Eriogonum* (Fig. 7a) contributed to the fast rate of spread.

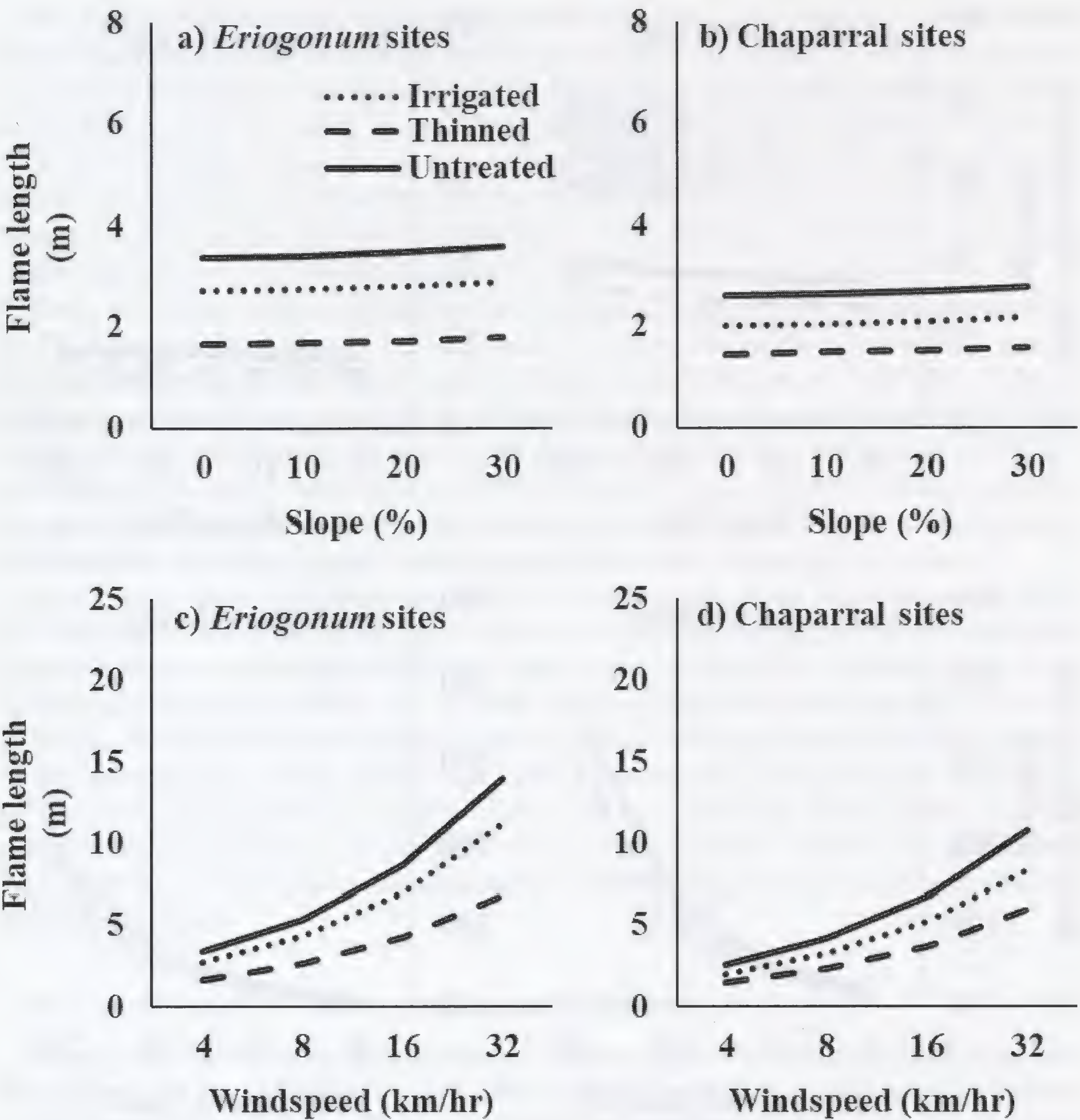


Fig. 8. Predicted flame lengths for changes in slope given the conditions present at the a) *Eriogonum* and b) chaparral sites, and for changes in wind speed at the c) *Eriogonum* and d) chaparral sites.

Using these scenarios, we asked the question how changes in slope and wind speed would alter these patterns. For *Eriogonum* sites and chaparral sites under these conditions, it is apparent that slope had only a minimal impact on flame length (Fig. 8a & b). Windspeed on the other hand had a much greater impact on flame length and this was most pronounced in the finer fuels of the sage scrub *Eriogonum* (Fig. 8c). Rate of fire spread followed a similar pattern (Fig. 9) with slope playing far less of a role than wind speed.

Discussion and Conclusions

Reducing housing losses at the wildland-urban interface often requires clearance of vegetation around the structures, however, there is growing evidence that maintaining natural landscapes of green woody shrubs and trees can be compatible with reducing fire hazard (Gibbons et al. 2018). This is important because it provides homeowners options of

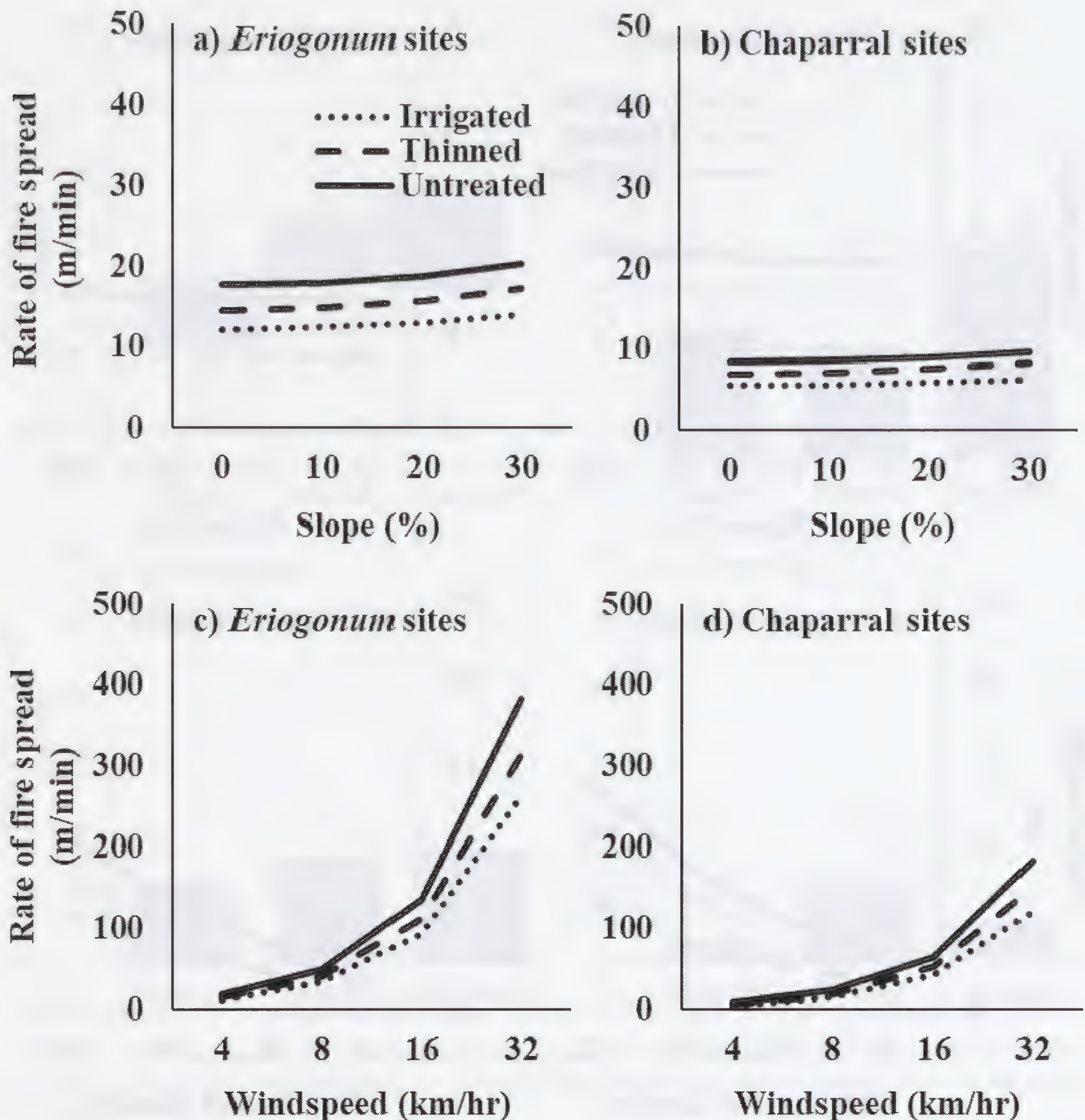


Fig. 9. Predicted rate of fire spread for changes in slope given the conditions present at the a) *Eriogonum* and b) chaparral sites, and for changes in wind speed at the c) *Eriogonum* and d) chaparral sites.

how they can produce esthetically pleasing landscapes around homes and yet still provide some level of fire protection. Greenness of vegetation around homes has been shown to be associated with a reduced loss from wildfires (Gibbons et al. 2018), and there are several possible explanations for this pattern. Greenbelts around home often have reduced levels of flammability but other possibilities include a potential role for green trees acting as ‘ember-catchers’ particularly in wind-driven fires (Keeley and Syphard 2019).

The present study examined greenbelts that utilized native shrubs that were the same species as surrounding wildland vegetation but were lightly irrigated during the summer drought. We showed that live fuel moisture during summer and fall were typically higher in these irrigated shrubs than in adjacent unirrigated shrubs of the same species and using these data fire behavior models suggested these native shrub landscapes provided a level of fire protection.

For all species, both sage scrub *Eriogonum fasciculatum*, and chaparral species, flame length and rate of spread were lower than untreated adjacent chaparral. When this vegetation was thinned rate of fire spread was still higher than the irrigated treatment in three of the four species. However, there was no improvement in flame lengths by irrigating these species. Thus, thinning natural vegetation has variable benefits in terms of reducing fire hazard and in some respects may be no different than lightly irrigated treatments. However, one important distinction is that homeowner preference of lightly irrigated green native vegetation during the summer over the dried thinned native vegetation.

There are clearly species differences in the role of irrigation in affecting summer-fall LFMC and these have impacts on fire behavior. Indeed, this study contributes to a need for studies that link LFMC to fire behavior (Weise et al. 1998; Pimont et al. 2019). Species differences observed in this study follow patterns observed by Pivovarovoff et al. (2019). Both studies showed that during the dry season *Malosma laurina* had the highest LFMC of all species tested and Pivovarovoff et al. (2019) showed this was linked to the least negative foliage water potentials. These patterns potentially are tied to the very deep root systems of this species that allows access to deep underground water during the dry season.

In summary, there is evidence that lightly irrigated native shrub landscaping can meet many desirable outcomes. Using native vegetation that has a long legacy on this landscape is potentially more compatible with the native fauna in the region (Jeschke et al. 2014, Tallamy and Shropshire 2009). For example, native plants can maintain greater diversity of insects that have direct and indirect benefits and can affect populations at other trophic levels (Narengo et al. 2018). Pollination webs are potentially greater as well as diversity of dispersers, both of which play important ecosystem processes. These lightly irrigated native landscapes consume far less water than more traditional landscaping (Rubin and Warren 2013) and these plants maintain higher LFMC during the dry season, contributing to reduced fire risk.

Acknowledgments

Support was provided by Department of Defense, NAVFAC SW and Naval Base San Diego Contract No. N62473-14-2-0006. We thank Anne Pfaff for assistance with figures, and Taylor Johnson and Lisa Markovchick with fuel moisture sampling. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Literature Cited

- Countryman, C.M. and W.A. Dean. 1979. Measuring moisture content in living chaparral: a field user's manual. Berkeley, California, USDA Forest Service, General Technical Report PSW-36, Pacific Southwest Forest and Range Experimental Station.
- Cox, G. 1990. Laboratory manual of general ecology, 6th ed. William C. Brown.
- Dennison, P.E., M.A. Moritz and R.S. Taylor. 2008. Evaluating predictive models of critical live fuel moisture in the Santa Monica Mountains, California. *Int. J. Wildl. Fire*, 17:18-27.
- Dennison, P.E. and M.A. Moritz. 2010. Critical live fuel moisture in chaparral ecosystems: A threshold for fire activity and its relationship to antecedent precipitation. *Int. J. Wildl. Fire*, 18:1021-1027.
- Gardner, P.D., H.J. Corner and K. Widaman. 1987. The risk perceptions and policy response toward wildland fire hazards by urban home-owners. *Landscape and Urban Planning*, 14:163-172.
- Gibbons, P., A.M. Gill, N. Shopre, M.A. Moritz, S. Dovers and G.J. Cary. 2018. Options for reducing house-losses during wildfires without clearing trees and shrubs. *Landscape and Urban Planning*, 174:10-17.

- Haase, S.M., J.J. Sanchez and D.R. Weise. 2016. Evaluation of standard methods for collecting and processing fuel moisture samples. Albany, CA, USDA Forest Service. Research Paper PSW-RP-268: 28.
- Jeschke, J.M., S. Bacher, T.M. Blackburn, J.T.A. Dick, F. Ess, T. Evans, M. Gaertner, P.E. Hulme, I. Kuhn, A. Mrugala, W. Perql, A. Ricciardi, M.M. Richardson, A. Sendek, M. Vila, M. Winter and S. Kumschick. 2014. Defining the impact of non-native species. *Cons. Biol.*, 28:1188-1194.
- Keeley, J.E. and A.D. Syphard. 2019. Twenty-first century California, USA, wildfires: fuel-dominated vs. wind-dominated fires. *Fire Ecology*, 15:24: doi.org/10.1186/s42408-019-0041-0.
- Kent, D. 2005. Creating fire-resistant landscapes, gardens, and properties in California's diverse environments, Wilderness.
- Narango, D.L., D.W. Tallamy and P.P. Marra. 2018. Nonnative plants reduce population growth of an insectivorous bird. *PNAS*, 115:11549-11554.
- Pellizzoro, G., A. Duce and P. Zara. 2007. Seasonal variations of live moisture content and ignitability in shrubs of the Mediterranean Basin. *Int. J. Wildl. Fire*, 16:633-641.
- Penman, S.H., O.F. Price, T.D. Penman and R.A. Bradstock. 2019. The role of defensible space on the likelihood of house impact from wildfires in forested landscapes of south eastern Australia. *Int. J. Wildl. Fire*, 28:4-14.
- Pimont, F., J. Ruffault, N. Martin-St Paul, J.L. Dupuy. 2019. Why is the effect of live fuel moisture content on fire rate of spread underestimated in field experiments in shrublands? *Int. J. Wildl. Fire*, 28:127-137.
- Pivovarovoff, A.L., N. Emery, M.R. Sharifi, M. Witter, J.E. Keeley and P.W. Rundel. 2019. The effect of ecophysiological traits on live fuel moisture content. *Fire*, 22:12.
- Pollet, J. and A. Brown. 2007. Fuel Moisture Sampling Guide. Salt Lake City, Utah, Bureau of Land Management, Utah State Office: 32.
- Rossa, C.G. and P.M. Fernandes. 2018. Live fuel moisture content: The 'pea under the mattress' of fire spread rate modeling? *Fire*, 13:43: <https://doi.org/10.3390/fire1030043>.
- Rothermel, R.C. 1983. How to predict the spread and intensity of forest and range fires, USDA Forest Service, Intermountain Forest and Range Experiment Station, Ogden UT 84401. INT-143, 161.
- Rubin, G. and L. Warren. 2013. *The California Native Landscape*, Timber Press.
- Stow, D., M. Niphadkar and J. Kaiser. 2005. MODIS-derived visible atmospherically resistant index for monitoring chaparral moisture content. *Int. J. Remote Sens.*, 26:3867-3873.
- Syphard, A.D., T.J. Brennan, and J.E. Keeley. 2014. The role of defensible space. J. for residential structure protection during wildfires. *Int. J. Wildl. Fire*, 23:1165-1175.
- Tallamy, D.W. and K. Shropshire. Ranking lepidopteran use of native versus introduced plants. *Cons. Bio.*, 23:941-947.
- Weise, D.R., R.A. Hartford and L. Mahaffey. 1998. Assessing live fuel moisture for fire management applications. Pp 49-55 in *Fire in Ecosystem Management: Sifting the Paradigm from Suppression to Prescription*, Tall Timbers Fire Ecology Conference Proceedings No. 20 (T.L. Pruden and L.A. Brennan, eds.). Tall Timbers Research Station.

Appendix I. Structural data for fire modeling for the three treatments at the three sites for both chaparral and sage scrub species.

Site	Treatment	Vegetation type	% veg. cover	Average height (m)	% of veg. live in primary layer	Live fuel moisture (species)	Dead fuel moisture
Highcountry	Control	Chaparral	90	1.8	75	Arcgla	1, 10, 100 hr fuels - 9%, 10%, 11%
Highcountry	Thinned	Chaparral	28	1.5	94	Arcgla	1, 10, 100 hr fuels - 9%, 10%, 11%
Highcountry	Planted & irrigated	Chaparral	90	1.8	75	Arcgla	1, 10, 100 hr fuels - 9%, 10%, 11%
Highcountry	Control	Sage scrub	98	1.4	90	Erofas	1, 10, 100 hr fuels - 9%, 10%, 11%
Highcountry	Thinned	Sage scrub	41	1	100	Erofas	1, 10, 100 hr fuels - 9%, 10%, 11%
Highcountry	Planted & irrigated	Sage scrub	98	1.4	90	Erofas	1, 10, 100 hr fuels - 9%, 10%, 11%
Oak Grove	Control	Chaparral	88	1.5	94	Mallau	1, 10, 100 hr fuels - 9%, 10%, 11%
Oat Grove	Thinned	Chaparral	50	1.5	95	Mallau	1, 10, 100 hr fuels - 9%, 10%, 11%
Oak Grove	Planted & irrigated	Chaparral	88	1.4	94	Mallau	1, 10, 100 hr fuels - 9%, 10%, 11%
Oak Grove	Control	Sage scrub	91	1.1	100	Erofas	1, 10, 100 hr fuels - 9%, 10%, 11%
Oak Grove	Thinned	Sage scrub	29	1	65	Erofas	1, 10, 100 hr fuels - 9%, 10%, 11%
Oak Grove	Planted & irrigated	Sage scrub	91	1.1	100	Erofas	1, 10, 100 hr fuels - 9%, 10%, 11%
Rangeland	Control	Chaparral	97	2.5	91	Queber	1, 10, 100 hr fuels - 9%, 10%, 11%
Rangeland	Thinned	Chaparral	58	1.9	75	Queber	1, 10, 100 hr fuels - 9%, 10%, 11%
Rangeland	Planted & irrigated	Chaparral	97	2.4	91	Queber	1, 10, 100 hr fuels - 9%, 10%, 11%
Rangeland	Control	Sage scrub	95	0.8	65	Erofas	1, 10, 100 hr fuels - 9%, 10%, 11%
Rangeland	Thinned	Sage scrub	75	0.8	66	Erofas	1, 10, 100 hr fuels - 9%, 10%, 11%
Rangeland	Planted & irrigated	Sage scrub	95	0.7	65	Erofas	1, 10, 100 hr fuels - 9%, 10%, 11%



CONTENTS

Survival and Recruitment of Rehabilitated Caspian Terns in Southern California Julie M. Skoglund, Rebecca S. Duerr, and Charles T. Collins.....	1
Mitigation Ponds Offer Drought Resiliency for Western Spadefoot (<i>Spea hammondi</i>) Populations Katherine L. Baumberger, Adam R. Backlin, Elizabeth A. Gallegos, Cynthia J. Hitchcock, and Robert N. Fisher.....	6
Are Fishes Attracted to Piers? Movements and Association of Marine Fishes to a Public Fishing Pier within a Commercial Harbor Armand A. Barilotti, Connor F. White, and Christopher G. Lowe	18
Protecting the Wildland-Urban Interface in California: Greenbelts vs Thinning for Wildfire Threats to Homes Jon E. Keeley, Greg Rubin, Teresa Brennan, and Bernadette Piffard.....	35